Part III

7 Analysis of Corridor Condition

Section 7.A: Hydrologic and Hydraulic Processes

Understanding how water flows into and through stream corridors is critical to developing restoration initiatives. How fast, how much, how deep, how often, and when water flows are important basic questions that must be answered in order to make appropriate decisions about the implementation of a stream corridor's restoration.

Section 7.B: Geomorphic Processes

This section combines the basic hydrologic processes with the physical or geomorphic functions and characteristics. Water flows through streams but is affected by the kinds of soils and alluvial features within the channel, in the floodplain, and in the uplands. The amount and kind of sediments carried by a stream is largely a determinant of its equilibrium characteristics, including size, shape, and profile. Successful implementation of the stream corridor restoration, whether active (requiring direct intervention) or passive, (removing only disturbance factors), depends on an understanding of how water and sediment are related to channel form and function, and on what processes are involved with channel evolution.

Section 7.C: Physical and Chemical Characteristics

The quality of water in the stream corridor is normally a primary objective of restoration, either to improve it to a desired condition, or to sustain it. Restoration initiatives should consider the physical and chemical characteristics that may not be readily apparent but that are nonetheless critical to the functions and processes of stream corridors. Chemical manipulation of specific characteristics usually involves the management or alteration of elements in the landscape or corridor.

Section 7.D: Biological Community Characteristics

The fish, wildlife, plants, and human beings that use the stream corridor, live in, or just visit the stream corridor are key elements to consider, not only in terms of increasing *populations or species diversity, but also in terms of usually being one of the primary goals of the restoration effort. A thorough understanding of how water flows, how sediment is transported, and how geomorphic features and processes evolve is important. However, a prerequisite to successful restoration is an understanding of the living parts of the system and how the physical and chemical processes affect the stream corridor.*

7.A. Hydrologic Processes

Flow Analysis

Restoring stream structure and function requires knowledge of flow characteristics. At a minimum, it is helpful to know whether the stream is perennial, intermittent, or ephemeral, and the relative contributions of baseflow and stormflow in the annual runoff. It might also be helpful to know whether streamflow is derived primarily from rainfall, snowmelt, or a combination of the two.

Other desirable information includes the relative frequency and duration of extreme high and low flows for the site and the duration of certain stream flow levels. High and low flow extremes usually are described with a statistical procedure called a frequency analysis, and the amount of time that various flow levels are present is usually described with a flow duration curve.

Finally, it is often desirable to estimate the channel-forming or dominant discharge for a stream (i.e., the discharge that is most effective in shaping and maintaining the natural stream channel). *Channel-forming* or *dominant discharge* is used for design when the restoration includes channel reconstruction.

Estimates of streamflow characteristics needed for restoration can be obtained from stream gauge data. Procedures for determining flow duration characteristics and the magnitude and frequency of floods and low flows at gauged sites are described in this section. The procedures are illustrated using daily mean flows and annual peak flows (the maximum discharge for each year) for the Scott River near Fort Jones, a 653-square-mile watershed in northern California.

Most stream corridor restoration initiatives are on streams or reaches that lack systematic stream gauge data. Therefore, estimates of flow duration and the frequency of extreme high and low flows must be based on indirect methods from regional hydrologic analysis. Several methods are available for indirect estimation of mean annual flow and flood characteristics; however, few methods have been developed for estimating low flows and general flow duration characteristics.

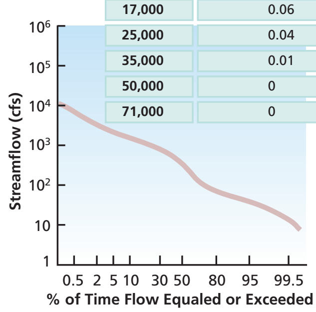
Users are cautioned that statistical analyses using historical streamflow data need to account for watershed changes that might have occurred during the period of record. Many basins in the United States have experienced substantial urbanization and development; construction of upstream reservoirs, dams, and storm water management structures; and construction of levees or channel modifications. These features have a direct impact on the statistical analyses of the data for peak flows, and for low flows and flow duration curves in some instances. Depending on basin modifications and the analyses to be performed, this could require substantial time and effort.

Flow Duration

The amount of time certain flow levels exist in the stream is represented by a

Figure 7.1: Flow duration curve and associated data tables
Data for the Scott River, near Fort Jones, CA, 1951-1980, show that a flow of 1,100 cubic feet per second (cfs) is exceeded about 20 percent of the time.
From Lumb et al. (1990).

Discharges (cfs)	% of Time Flow Equaled or Exceeded
0	100
1	100
1.4	100
2	100
2.8	100
4	100
5.7	99.96
8.1	99.76
11	99.68
16	99.43
23	98.7
33	96.89
46	94.2
66	85.02
93	74.54
130	65.98
190	60.15
270	55.03
380	49.03
530	42.05
760	31.41
1,100	20.75
1,500	11.95
2,200	5.1
3,100	2.25
4,300	1.2
6,100	0.68
8,700	0.35
12,000	0.16
17,000	0.06
25,000	0.04
35,000	0.01
50,000	0
71,000	0



flow duration curve which depicts the percentage of time a given streamflow was equaled or exceeded over a given period. Flow duration curves are usually based on daily streamflow (a record containing the average flow for each day) and describe the flow characteristics of a stream throughout a range of discharges without regard to the sequence of occurrence. A flow duration curve is the cumulative histogram of the set of all daily flows. The construction of flow duration curves is described by Searcy (1959), who recommends defining the cumulative histogram of streamflow by using 25 to 35 well-distributed class intervals of streamflow data.

Figure 7.1 is a flow duration curve that was defined using 34 class intervals and software documented by Lumb et al. (1990). The numerical output is provided in the accompanying table.

The curve shows that a daily mean flow of 1,100 cubic feet per second (cfs) is exceeded about 20 percent of the time or by about 20 percent of the observed daily flows. The long-term mean daily flow (the average flow for the period of record) for this watershed was determined to be 623 cfs. The duration curve shows that this flow is exceeded about 38 percent of the time.

For over half the states, the USGS has published reports for estimating flow duration percentiles and low flows at ungauged locations. Estimating flow duration characteristics at ungauged sites usually is attempted by adjusting data from a nearby stream gauge in a hydrologically similar basin. Flow duration characteristics from the stream gauge record are expressed per unit area of drainage basin at the gauge (i.e., in cfs/mi^2) and are multiplied by the drainage area of the ungauged site to estimate flow duration characteristics there. The accuracy of such a procedure is directly related to the similarity of the two sites. Generally, the drainage area at the stream gauge and ungauged sites should be fairly similar, and streamflow characteristics should be similar for both sites. Additionally, mean basin elevation and physiography should be similar for both sites. Such a procedure does not work well and should not be attempted in stream systems dominated by local convective storm runoff or where land uses vary significantly between the gauged and ungauged basins.

Flow Frequency Analysis

The frequency of floods and low flows for gauged sites is determined by analyzing an annual time series of maximum or minimum flow values (a chronological list of the largest or smallest flow that occurred each year). Although previously described in Chapter 1, *flow frequency* is redefined here because of its relevance to the sections that follow. Flow frequency is defined as the probability or percent chance of a given flow's being exceeded or not exceeded in any given year. Flow frequency is often expressed in terms of *recurrence interval* or the average number of years between exceeding or not exceeding the given flows. For example, a given flood flow that has a 100-year recurrence interval is expected to be exceeded, on average, only once in any 100-year period; that is, in any given year, the annual flood flow has a 1 percent chance or 0.01 probability of exceeding the 100-year flood. The exceedance probability, p, and the recurrence interval, T, are related in that one is the reciprocal of the other (i.e., $T = 1/p$). Statistical procedures for determining the frequency of floods and low flows at gauged sites follow.

As mentioned earlier, most stream corridor restoration initiatives are on streams or reaches lacking systematic stream gauge data; therefore, estimates of flow duration characteristics and the frequency of extreme high and extreme low flows must be based on indirect methods from regional hydrologic analysis.

Flood Frequency Analysis

Guidelines for determining the frequency of floods at a particular location using streamflow records are documented by the Hydrology Subcommittee of the Interagency Advisory Committee on Water Data (IACWD 1982, Bulletin 17B). The guidelines described in Bulletin 17B are used by all federal agencies in planning activities involving water and related land resources. Bulletin 17B recommends fitting the Pearson Type III frequency distribution to the logarithms of the annual peak flows using sample statistics (mean, standard deviation, and skew) to estimate the distribution parameters. Procedures for outlier detection and adjustment, adjustment for historical data, development of generalized skew, and weighting of station and generalized skews are provided. The station skew is computed from the observed peak flows, and the generalized skew is a regional estimate determined from estimates at several long-term stations in the region. The US Army Corps of Engineers also has produced a user's manual for *flood frequency analysis* (Report CPD-13, 1994) that can aid in determining flood frequency distribution parameters. NRCS has also produced a manual (*National Engineering Handbook*, Section 4, Chapter 18) that can also be used in determining flood frequency distribution (USDA-SCS 1983).

Throughout the United States, flood frequency estimates for USGS gauging

Sources of Daily Mean Discharge and Other Data from USGS Stream Gauges

Daily Mean Streamflow

Daily mean streamflow data needed for defining flow duration curves are published on a water-year (October 1 to September 30) basis for each state by the U.S. Geological Survey (USGS) in the report series Water Resources Data. The data collected and published by the USGS are archived in the National Water Information System (NWIS).

The USGS currently provides access to streamflow data by means of the Internet. The USGS URL address for access to streamflow data is http:/water.usgs.gov. Approximately 400,000 station years of historical daily mean flows for about 18,500 stations are available through this source. The USGS data for the entire United States are also available from commercial vendors on two CD-ROMs, one for the eastern and one for the western half of the country (e.g., CD-ROMs for DOS can be obtained from Earth Info, and CD-ROMs for Windows can be obtained from Hydrosphere Data Products. Both companies are located in Boulder, Colorado.)

In addition to the daily mean flows, summary statistics are also published for active streamflow stations in the USGS annual Water Resources Data reports. Among the summary statistics are the daily mean flows that are exceeded 10, 50, and 90 percent of the time of record. These durations are computed by ranking the observed daily mean flows from q_i to $q_{(n \bullet 365)}$ where n is the number of years of record, $q_{(1)}$ is the largest observation, and $q_{(365 \bullet n)}$ is the smallest observation. The ranked list is called a set of ordered observations. The $q_{(1)}$ that are exceeded 10, 50, and 90 percent of time are then determined. Flow duration percentiles (quantiles) for gauged sites are also published by USGS in reports on low flow frequency and other streamflow statistics (e.g., Atkins and Pearman 1994, Zalants 1991, Telis 1991, and Ries 1994).

Peak Flow

Annual *peak flow* data needed for flood frequency analysis are also published by the USGS, archived in NWIS, and available through the internet at the URL address provided above. Flood frequency estimates at gauged sites are routinely published by USGS as part of cooperative studies with state agencies to develop regional regression equations for ungauged watersheds. Jennings et al. (1994) provide a nationwide summary of the current USGS reports that summarize flood frequency estimates at gauged sites as well as regression equations for estimating flood peak flows for ungauged watersheds. Annual and partial-duration (peaks-above-threshold) peak flow data for all USGS gauges can be obtained on one CD-ROM from commercial vendors.

stations have been correlated with certain climatic and basin characteristics. The result is a set of regression equations that can be used to estimate flood magnitude for various return periods in ungauged basins (Jennings et al. 1994). Reports outlining these equations often are prepared for state highway departments to help them size culverts and rural road bridge openings.

Estimates of the frequency of peak flows at ungauged sites may be made by using these regional regression equations, provided that the gauged and ungauged sites have similar climatic and physiographic characteristics.

Frequently the user needs only such limited information as mean annual precipitation, drainage area, storage in lakes and wetlands, land use, major soil types, stream gradients, and a topographic map to calculate flood magnitudes at a site. Again, the accuracy of the procedure is directly related to the hydrologic similarity of the two sites.

Similarly, in many locations, flood frequency estimates from USGS gauging stations have been correlated with certain channel geometry characteristics. These correlations produce a set of regression equations relating some channel feature, usually active channel width, to flood magnitudes for various return periods. A review of these equations is provided by Wharton (1995). Again, the standard errors of the estimate might be large.

Regardless of the procedure or source of information chosen for obtaining flood frequency information, estimates for the 1.5, 2, 5, 10, 25, and (record permitting) 50 and 100-year flood events may be plotted on standard log-probability paper, and a smooth curve may be drawn between the points. (Note that these are flood events with probabilities of 67, 50, 20, 10, 4, 2, and 1 percent, respectively.) This plot becomes the flood frequency relationship for the restoration site under consideration. It provides the background information for determining the frequency of inundation of surfaces and vegetation communities along the channel.

Flood Frequency Estimates

Flood frequency estimates also may be generated using precipitation data and applicable watershed runoff models such as HEC-1, TR-20, and TR-55. The precipitation record for various return-period storm events is used by the watershed model to generate a runoff hydrograph and peak flow for that event. The modeled rainfall may be from historical data or from an assumed time distribution of precipitation (e.g., a 2-year, 24-hour rainfall event). This method of generating flood frequency estimates assumes the return period of the runoff event equals the return period of the precipitation event (e.g., a 2-year rainfall event will generate a 2-year peak flow). The validity of this assumption depends on antecedent moisture conditions, basin size, and a number of other factors.

Low-Flow Frequency Analysis

Guidelines for *low-flow frequency analysis* are not as standardized as those for flood frequency analysis. No single frequency distribution or curve-fitting method has been generally accepted. Vogel and Kroll (1989) provide a summary of the limited number of studies that have evaluated frequency distributions and fitting methods for low flows. The methodology used by USGS and USEPA is described below.

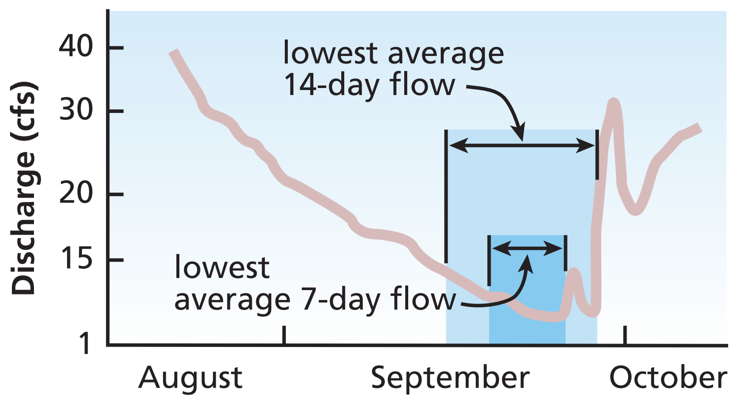


Figure 7.2: Annual hydrograph displaying low flows.
The daily mean flows on the lowest part of the annual hydrograph are averaged to give the 7-day and 14-day low flows for that year.

The hypothetical daily hydrograph shown in **Figure 7.2** is typical of many areas of the United States where the annual minimum flows occur in late summer and early fall. The climatic year (April 1 to March 31) rather than the water year is used in low-flow analyses so that the entire low-flow period is contained within one year.

Data used in low-flow frequency analyses are typically the annual minimum average flow for a specified number of consecutive days. The annual minimum 7- and 14-day low flows are illustrated in Figure 7.2. For example, the annual minimum 7-day flow is the annual minimum value of running 7-day means.

USGS and USEPA recommend using the Pearson Type III distribution to the logarithms of annual minimum d-day low flows to obtain the flow with a nonexceedance probability p (or recurrence interval T = 1/p). The Pearson Type III low-flow estimates are computed from the following equation:

$$X_{d,T} = M_d - K_T S_d$$

where:

$X_{d,T}$ = the logarithm of the annual minimum d-day low flow for which the flow is not exceeded in 1 of T years or which has a probability of p = 1/T of not being exceeded in any given year

M_d = the mean of the logarithms of annual minimum d-day low flows

S_d = the standard deviation of the logarithms of the annual minimum d-day low flows

K_T = the Pearson Type III frequency factor

The desired quantile, $Q_{d,T}$, can be obtained by taking the antilogarithm of the equation.

The 7-day, 10-year low flow ($Q_{7,10}$) is used by about half of the regulatory agencies in the United States for managing water quality in receiving waters (USEPA 1986, Riggs et al. 1980). Low flows for other durations and frequencies are used in some states.

Computer software for performing low-flow analyses using a record of daily mean flows is documented by Hutchison (1975) and Lumb et al. (1990). An example of a low-flow frequency curve for the annual minimum 7-day low flow is given in **Figure 7.3** for Scott River near Fort Jones, California, for the same period (1951 to 1980) used in the flood frequency analyses above.

From Figure 7.3, one can determine that the $Q_{7,10}$ is about 20 cfs, which is comparable to the 99th percentile (daily mean flow exceeded 99 percent of the time) of the flow duration curve (Figure 7.1). This comparison is consistent with findings of Fennessey and Vogel (1990), who concluded that the $Q_{7,10}$ from 23 rivers in Massachusetts was approximately equal to the 99th flow duration percentile. The USGS routinely publishes low flow estimates at gauged sites (Zalants 1991, Telis 1991, Atkins and Pearman 1994).

Following are discussions of different ways to look at the flows that tend to form and maintain streams. Restorations that include alterations of flows or changes in the dimensions of the stream must include engineering analyses as described in Chapter 8.

Channel-forming Flow

The *channel-forming* or *dominant discharge* is a theoretical discharge that if constantly maintained in an alluvial stream over a long period of time would produce the same channel geometry that is produced by the long-term natural hydrograph. Channel-forming discharge is the most commonly used single independent variable that is found to govern channel shape and form. Using a channel-forming discharge to design channel geometry is not a universally accepted technique, although most river engineers and scientists agree that the

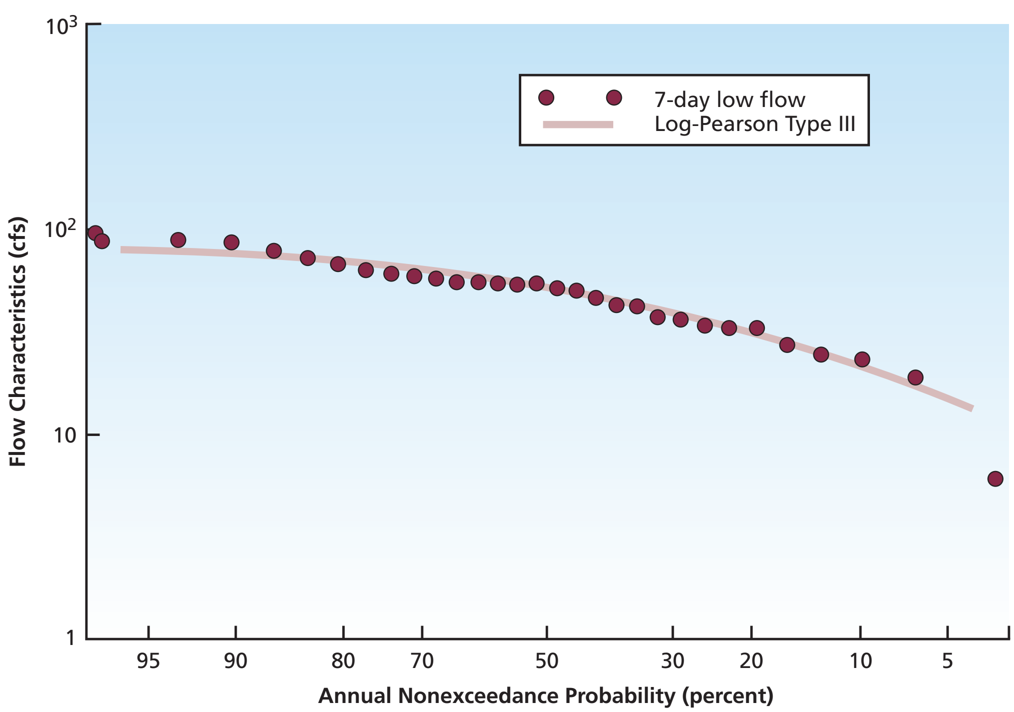


Figure 7.3: Annual minimum 7-day low flow frequency curve. *The $Q_{7,10}$ on this graph is about 20 cfs. The annual minimum value of 7-day running means for this gauge is about 10 percent.*

concept has merit, at least for perennial (humid and temperate) and perhaps ephemeral (semiarid) rivers. For arid channels, where runoff is generated by localized high-intensity storms and the absence of vegetation ensures that the channel will adjust to each major flood event, the channel-forming discharge concept is generally not applicable.

Natural alluvial rivers experience a wide range of discharges and may adjust their geometry to flow events of different magnitudes by mobilizing either bed or bank sediments. Although Wolman and Miller (1960) noted that "it is logical to assume that the channel shape is affected by a range of flows rather than a single discharge," they concurred with the view put forward earlier by civil engineers working on "regime theory" that the channel-forming or dominant discharge is the steady flow that produces the same gross channel shapes and dimensions as the natural sequence of events (Inglis 1949). Wolman and Miller (1960) defined "moderate frequency" as events occurring "at least once each year or two and in many cases several or more times per year." They also considered the sediment load transported by a given flow as a percentage of the total amount of sediment carried by the river during the period of record. Their results, for a variety of American rivers located in different climatic and physiographic regions, showed that the greater part (that is, 50 percent or more) of the total sediment load was carried by moderate flows rather than catastrophic floods. Ninety percent of the load was carried by events with a return period of less than 5 years. The precise form of the cumulative curve actually depends on factors such as the predominant mode of transport (bed load, suspended load, or mixed load) and the flow variability, which is influenced by the size and hydrologic characteristics of the watershed. Small watersheds generally experience a wider range of flows than large watersheds, and this tends to increase the proportion of sediment load carried by infrequent events. Thorough reviews of arguments about the conceptual basis of channel-forming discharge theory can be found in textbooks by Richards (1982), Knighton (1984), and Summerfield (1991).

Researchers have used various discharge levels to represent the channel-forming discharge. The most common are (1) bankfull discharge, (2) a specific discharge recurrence interval from the annual peak or partial duration frequency curves, and (3) effective discharge. These approaches are frequently used and can produce a good approximation of the channel-forming discharge in many situations; however, as discussed in the following paragraphs, considerable uncertainties are involved in all three of these approaches. Many practitioners are using specific approaches to determine channel-forming discharge and the response of stream corridors. Bibliographic information on these methods is available later in the document.

Because of the spatial variability within a given geographical region, the response of any particular stream corridor within the region can differ from that expected for the region as a whole. This is especially critical for streams draining small, ungauged drainage areas. Therefore, the ex-

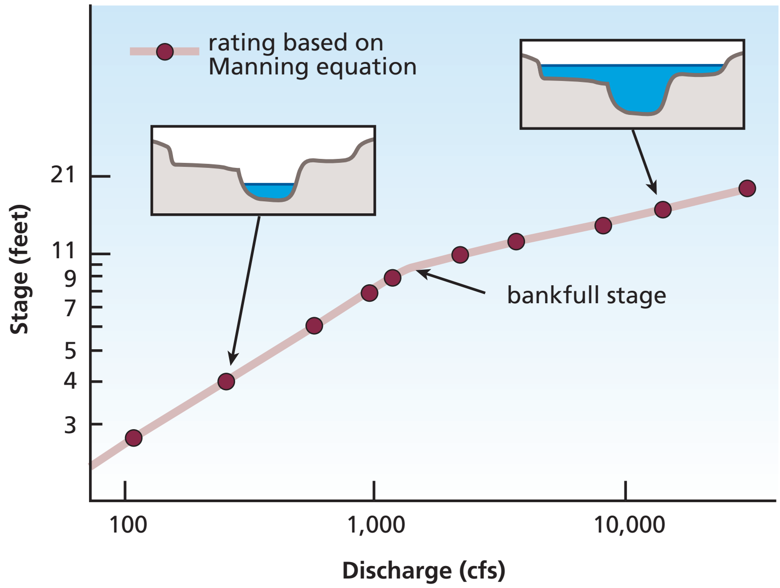


Figure 7.4: Determination of bankfull stage from a rating curve.
The discharge that corresponds to the elevation of the first flat depositional surface is the bankfull discharge.

pected channel-forming discharge of ungauged areas should be estimated by more than one alternative method, hopefully leading to consistent estimates.

Bankfull Discharge

The *bankfull discharge* is the discharge that fills a stable alluvial channel up to the elevation of the active floodplain. In many natural channels, this is the discharge that just fills the cross section without overtopping the banks, hence the term "bankfull." This discharge is considered to have morphological significance because it represents the breakpoint between the processes of channel formation and floodplain formation. In stable alluvial channels, bankfull discharge corresponds closely with effective discharge and channel-forming discharge.

The stage vs. discharge or rating curve presented in **Figure 7.4** was developed for a hypothetical stream by computing the discharge for different water surface elevations or stages. Since discharges greater than bankfull spread across the active floodplain, stage increases more gradually with increasing discharge above bankfull than below bankfull, when flows are confined to the channel. Another method for determining the bankfull stage and discharge is to determine the minimum value on a plot relating water surface elevation to the ratio of surface width to area. The frequency of the bankfull discharge can be determined from a frequency distribution plot like Figure 7.1.

Bankfull stage can also be identified from field indicators of the elevation of the active floodplain. The corresponding bankfull discharge is then determined from a stage vs. discharge relationship.

Field Indicators of Bankfull Discharge

Various field indicators can be used for estimating the elevation of the stage associated with bankfull flow. Although the first flat depositional surface is often used, the identification of depositional surfaces in the field can be difficult and misleading and, at the very least, requires trained, experienced field personnel. After an elevation is selected as the bankfull, the stage vs. discharge curve can be computed to determine the magnitude of the discharge corresponding to that elevation.

The above relationships seldom work in incised streams. In an incised stream, the top of the bank might be a terrace (an abandoned floodplain), and indicators of the active floodplain might be found well below the existing top of bank. In this situation, the elevation of the channel-forming discharge will be well below the top of the bank. In addition, the difference between the ordinary use of the term "bankfull" and the geomorphic use of the term can cause major communication problems.

Field identification of bankfull elevation can be difficult (Williams 1978), but is usually based on a minimum width/depth ratio (Wolman 1955), together with the recognition of some discontinuity in the nature of the channel banks such as a change in its sedimentary or vegetative characteristics. Others have defined bankfull discharge as follows:

- Nixon (1959) defined the bankfull stage as the highest elevation of a river that can be contained within the channel without spilling water on the river floodplain or washlands.
- Wolman and Leopold (1957) defined bankfull stage as the elevation of the active floodplain.
- Woodyer (1968) suggested bankfull stage as the elevation of the middle bench of rivers having several overflow surfaces.
- Pickup and Warner (1976) defined bankfull stage as the elevation at which the width/depth ratio becomes a minimum.

Bankfull stage has also been defined using morphologic factors, as follows:

- Schumm (1960) defined bankfull stage as the height of the lower limit of perennial vegetation, primarily trees.
- Similarly, Leopold (1994) states that bankfull stage is indicated by a change in vegetation, such as herbs, grasses, and shrubs.
- Finally, the bankfull stage is also defined as the average elevation of the highest surface of the channel bars (Wolman and Leopold 1957).

The field identification of bankfull stage indicators is often difficult and subjective and should be performed in stream reaches that are stable and alluvial (Knighton 1984). Additional guidelines are reviewed by Wharton (1995). In unstable streams, bankfull indicators are often missing, embryonic, or difficult to determine.

Direct determination of the discharge at bankfull stage is possible if a stream gauge is located near the reach of interest. Otherwise, discharge must be calculated using applicable hydraulic resistance equations and, preferably, standard hydraulic backwater techniques. This approach typically requires that an estimation of channel roughness be made, which adds to the uncertainty associated with calculated bankfull discharge.

Because of its convenience, bankfull discharge is widely used to represent channel-forming discharge. There is no universally accepted definition of bankfull stage or discharge that can be consistently applied, has general application, and integrates the processes that create the bankfull dimensions of the river. The reader is cautioned that the indicators used to define the bankfull condition must be spelled out each time a bankfull discharge is used in a project plan or design.

Determining Channel-Forming Discharge from Recurrence Interval

To avoid some of the problems related to field determination of bankfull stage, the *channel-forming discharge* is often assumed to be represented by a specific *recurrence interval* discharge. Some researchers consider this representative discharge to be equivalent to the bankfull discharge. Note that "bankfull discharge" is used synonymously with "channel-forming discharge" in this document. The earliest estimate for channel-forming discharge was the mean annual flow (Leopold and Maddock 1953). Wolman and Leopold (1957) suggested that the channel-forming discharge has a recurrence interval of 1 to 2 years. Dury (1973) concluded that the channel-forming discharge is approximately 97 percent of the 1.58-year discharge or the most probable annual flood. Hey (1975) showed that for three British gravel-bed rivers, the 1.5-year flow in an annual maximum series passed through the scatter of bankfull discharges measured along the course of the rivers. Richards (1982) suggested that in a partial duration series bankfull discharge equals the most probable annual flood, which has a 1 year return period. Leopold (1994) stated that most investigations have concluded that the bankfull discharge recurrence intervals ranged from 1.0 to 2.5 years. Pickup and Warner (1976) determined bankfull recurrence intervals ranged from 4 to 10 years on the annual series.

However, there are many instances where the bankfull discharge does not fall within this range. For example, Williams (1978) determined that approximately 75 percent of 51 streams that he analyzed appeared to have recurrence intervals for the bankfull discharge of between 1.03 and 5.0 years. Williams used the elevation of the active floodplain or the valley flat, if no active floodplain was defined at a station, as the elevation of the bankfull surface in his analyses. He did not establish whether these streams were in equilibrium, so the validity of using the top of the streambank as the bankfull elevation is in question, especially for those stations with valley flats. This might explain the wide range (1.02 to 200 years) he reported for bankfull discharge return intervals for streams with valley flats as opposed to active floodplains. The range in return intervals for 19 of the 28 streams with active floodplains was from 1.01 to 32 years. Nine of the 28 streams had bankfull discharge recurrence intervals of less than 1.0 year. It should be noted that only 3 of those 28 streams had bankfull discharge recurrence intervals

The reader is cautioned that the indicators used to define the bankfull condition must be spelled out each time a bankfull discharge is used in a project plan or design.

greater than 4.8 years. About one-third of the active floodplain stations had bankfull discharges near the 1.5-year recurrence interval.

Although the assumption that the channel-forming flow has a recurrence interval of 1 to 3 years is sufficient for reconnaissance-level studies, it should not be used for design until verified through inspection of reference reaches, data collection, and analysis. This is especially true in highly modified streams such as in urban or mined areas, as well as ephemeral streams in arid and semiarid areas.

Effective Discharge

The *effective discharge* is defined as the increment of discharge that transports the largest fraction of the sediment load over a period of years (Andrews 1980). The effective discharge incorporates the principle prescribed by Wolman and Miller (1960) that the channel-forming discharge is a function of both the magnitude of the event and its frequency of occurrence. An advantage of using the effective discharge is that it is a calculated rather than field-determined value. The effective discharge is calculated by numerically integrating the flow duration curve and the sediment transport rating curve. A graphical representation of the relationship between sediment transport, frequency of the transport, and the effective discharge is shown in **Figure 7.5**. The peak of curve C marks the discharge that is most effective in transporting sediment and, therefore, does the most work in forming the channel.

For stable alluvial streams, effective discharge has been shown to be highly correlated with bankfull discharge. Of the various discharges related to channel morphology (i.e., dominant, bankfull, and effective discharges), effective discharge is the only one that can be computed directly. The effective discharge has morphological significance since it is the discharge that transports the bulk of the sediment.

The effective discharge represents the single flow increment that is responsible for transporting the most sediment over some time period. However, there is a range of flows on either side of the effective discharge that also carry a significant portion of the total annual sediment load.

Biedenharn and Thorne (1994) used a graphical relationship between the cumulative percentage of sediment transported and the water discharge to define a range of effective discharges responsible for the majority of the sediment transport on the Lower

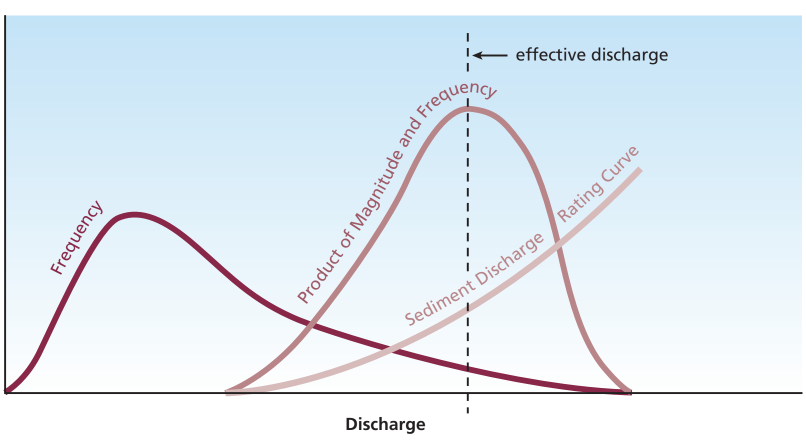


Figure 7.5: Effective discharge determination from sediment rating and flow duration curves. *The peak of curve C marks the discharge that is most effective in transporting sediment.* **From Wolman and Miller (1960).**

Mississippi River. They found that approximately 70 percent of the total sediment was moved in a range of flows between 500,000 cfs and 1,200,000 cfs, which corresponds to the flow that is equaled or exceeded 40 percent of the time and 3 percent of the time, respectively. Thorne et al. (1996) used a similar approach to define the range of effective discharges on the Brahmaputra River.

A standard procedure should be used for the determination of the effective discharge to ensure that the results for different sites can be compared. To be practical, it must either be based on readily available gauging station data or require only limited additional information and computational procedures.

The basic components required for calculation of effective discharge are (1) flow duration data and (2) sediment load as a function of water discharge. The method most commonly adopted for determining the effective discharge is to calculate the total bed material sediment load (tons) transported by each flow increment over a period of time by multiplying the frequency of occurrence for the flow increment (number of days) by the sediment load (tons/day) transported by that flow level. The flow increment with the largest product is the effective discharge. Although this approach has the merit of simplicity, the accuracy of the estimate of the effective discharge is clearly dependent on the calculation procedure adopted.

Values of mean daily discharges are usually used to compute the flow duration curve, as discussed above and presented in Figure 7.1. However, on flashy streams, mean daily values can underestimate the influence of the high flows, and, therefore, it might be necessary to reduce the discharge averaging period from 24 hours (mean daily) to 1 hour, or perhaps 15 minutes.

A *sediment rating curve* must be developed to determine the effective discharge. (See the *Sediment Yield and Delivery* section in Chapter 8 for more details.) The bed material load should be used in the calculation of the effective discharge. This sediment load can be determined from measured data or computed using an appropriate sediment transport equation. If measured suspended sediment data are used, the wash load should be subtracted and only the suspended bed material portion of the suspended load used. If the bed load is a significant portion of the load, it should be calculated using an appropriate sediment transport function and added to the suspended bed material load to provide an estimate of the total bed material load. If bed load measurements are available, these data can be used.

Determination of effective discharge using flow and sediment data is further discussed by Wolman and Miller (1960) and Carling (1988).

Determining Channel-Forming Discharge from Other Watershed Variables

When neither time nor resources permit field determination of bankfull discharge or data are unavailable to calculate the effective discharge,

Design Discharge and Ecological Function

Although a channel-forming or dominant discharge is important for design, it is often not sufficient for channel restoration initiatives. An assessment of a wider range of discharges might be necessary to ensure that the functional objectives of the project are met. For example, a restoration initiative targeting low-flow habitat conditions must consider the physical conditions in the channel during low flows.

indirect methods based on *regional hydrologic analysis* may be used (Ponce 1989). In its simplest form, regional analysis entails regression techniques to develop empirical relationships applicable to homogeneous hydrologic regions. For example, some workers have used watershed areas as surrogates for discharge (Brookes 1987a, Madej 1982, Newbury and Gaboury 1993). Regional relationships of drainage area with bankfull discharge can provide good starting points for selecting the channel-forming discharge.

Within hydrologically homogeneous regions where runoff varies with contributing area, runoff is proportional to watershed drainage area. Dunne and Leopold (1978) and Leopold (1994) developed average curves relating bankfull discharge to drainage area for widely separated regions of the United States. For example, relationships between bankfull discharge and drainage area for Brandywine Creek in Pennsylvania and the upper Green River basin in Wyoming are shown in the **Figure 7.6**.

Regional Relationship Between Bankfull and Mean Annual Discharge

Because the mean annual flow for each stream gauge operated by the USGS is readily available, it is useful to establish regional relationships between bankfull and mean annual discharges so that one can be estimated whenever the other is available. This information can be compared to the bankfull discharge estimated for any given ungauged site within a U.S. region. **The user is cautioned, however, that regional curve values have a high degree of error and can vary significantly for specific sites or reaches to be restored.**

Two important points are immediately apparent from Figure 7.6. First, humid regions that have sustained, widely distributed storms yield higher bankfull discharges per unit of drainage area than semiarid regions where storms of high intensity are usually localized. Second, bankfull discharge is correlated with drainage area, and the general relationship can be represented by functions of the form:

$$Q_{bf} = aA^b$$

where Q_{bf} is the bankfull discharge in cfs, A is the drainage area in square miles, and a and b are regression coefficients and exponents given in **Table 7.1**.

Establishing similar parametric relationships for other rivers of interest is useful because the upstream area draining into a stream corridor can be easily determined from either maps or digital terrain analysis tools. Once the area is determined, an estimate of the expected bankfull discharge for the corridor can be made from the above equation.

Mean Annual Flow

Another frequently used surrogate for channel-forming discharge in empirical regression equations is the *mean annual flow*. The mean annual flow, Q_m, is equivalent to the constant discharge that would yield the same volume of water in a water year as the sum of all continuously measured discharges. Just as in the case of bankfull discharge, Q_m varies proportionally with drainage area within hydrologically homogeneous basins. Given that both Q_{bf} and Q_m exhibit a similar functional dependence on A, a consistent proportionality is to be expected between these discharge measures within the same region. In fact, Leopold (1994) gives the following average values of the ratio Q_{bf}/Q_m for three widely separated regions of the United States: 29.4 for 21 stations in the Coast Range of California, 7.1 for 20 stations in the Front Range of Colorado, and 8.3 for 13 stations in the Eastern United States.

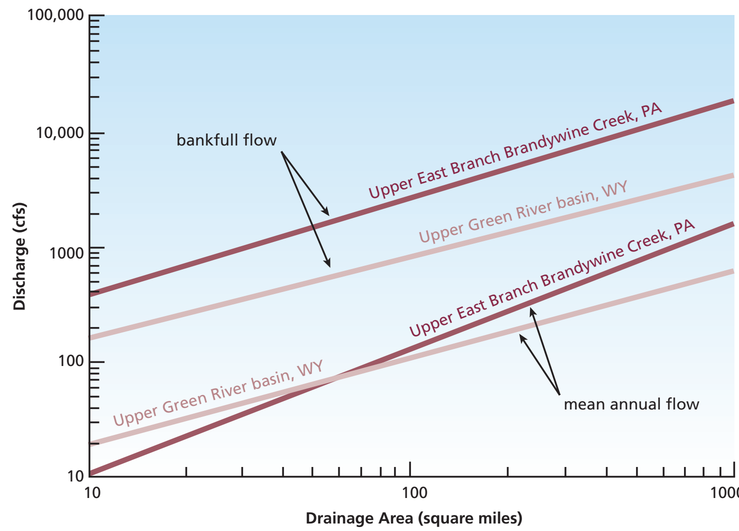


Figure 7.6: Regional relationships for bankfull and mean annual discharge as a function of drainage area.

The mean annual flow is normally less than the bankfull flow.

Source: Dunne and Leopold 1978.

Table 7.1: Functional parameters used in regional estimates of bankfull discharge.

In column a are regression coefficients and in column b are exponents that can be used in the bankfull discharge equation.

Source: Dunne and Leopold 1978.

River Basin	a	b
Southeastern PA	61	0.82
Upper Salmon River, ID	36	0.68
Upper Green River, ID	28	0.69
San Francisco Bay Region, CA	53	0.93

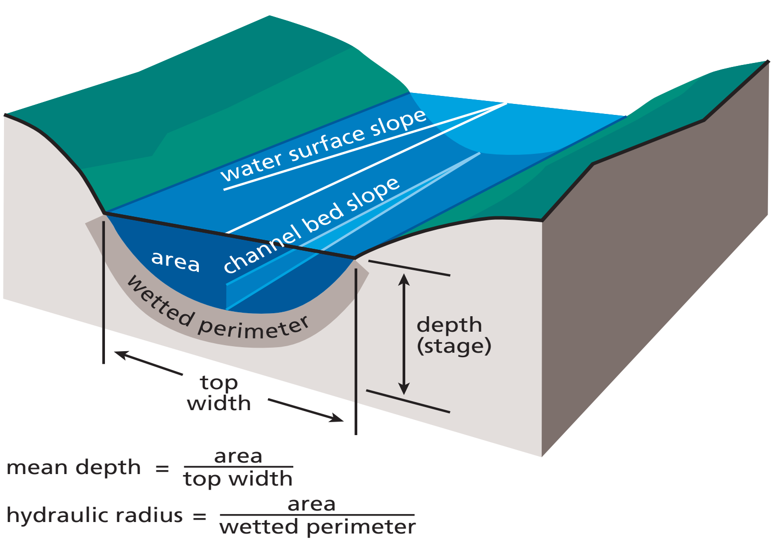
$Q_{bf} = aA^b$

Stage vs. Discharge Relationships

Surveys of stream channel cross sections are useful for analyzing channel form, function, and processes. Use of survey data to construct relationships among streamflow, channel geometry, and various hydraulic characteristics provides information that serves a variety of applications. Although stage-discharge curves often can be computed from such cross section data, users should be cautioned to verify their computations with direct discharge measurements whenever possible.

Information on stream channel geometry and hydraulic characteristics is useful for channel design, riparian area restoration, and instream structure placement. Ideally, once a channel-forming discharge is defined, the channel is designed to contain that flow and higher flows are allowed to spread over the floodplain. Such periodic flooding is extremely important for the formation of channel macrofeatures, such as point bars and meander bends, and for establishing certain kinds of riparian vegetation. A cross section analysis also may help in optimal design and placement of items such as culverts and fish habitat structures.

Figure 7.7: Hydraulic parameters.
Streams have specific cross-sectional and longitudinal profile characteristics.



Additionally, knowledge of the relationships between discharge and channel geometry and hydraulics is useful for reconstructing the conditions associated with a particular flow rate. For example, in many channel stability analyses, it is customary to relate movement of bed materials to some measure of stream power or average bed shear stress. If the relationships between discharge and certain hydraulic variables (e.g., mean depth and water surface slope) are known, it is possible to estimate stream power and average bed shear as a function of discharge. A cross section analysis therefore makes it possible to estimate conditions of substrate movement at various levels of streamflow.

Continuity Equation

Discharge at a cross section is computed using the simplified form of the *continuity equation*:

$$Q = AV$$

where:

Q = discharge

A = cross sectional area of the flow

V = average velocity in the downstream direction

Computing the cross-sectional area is a geometry problem. The area of interest is bounded by the channel cross section and the water surface elevation (stage) (**Figure 7.7**). In addition to cross-sectional area, the top width, wetted perimeter, mean depth, and hydraulic radius are computed for selected stages (Figure 7.7).

Uniform flow equations may be used for estimating mean velocity as a function of cross section hydraulic parameters.

Manning's Equation

Manning's equation was developed for conditions of uniform flow in which the water surface profile and energy grade line are parallel to the streambed, and the area, hydraulic radius, and average depth remain constant throughout the reach. The energy grade line is a theoretical line whose elevation above the streambed is the sum of the water surface elevation and a term that represents the kinetic energy of the flow (Chow 1959). The slope of the energy grade line represents the rate at which energy is dissipated through turbulence and boundary friction. When the water surface slope and the energy grade line parallel the streambed, the slope of the energy grade line is assumed to equal the water surface slope. When the slope of the energy grade line is known, various resistance formulas allow computing mean cross-sectional velocity.

The importance of Manning's equation in stream restoration is that it provides the basis for computing differences in flow velocities and elevations due to differences in hydraulic roughness. Note that the flow characteristics can be altered to meet the goals of the restoration either by direct intervention or by changing the vegetation and roughness of the stream. Manning's equation is also useful in determining bankfull discharge for bankfull stage.

Manning's equation is also used to calculate energy losses in natural channels with gradually varied flow. In this case, calculations proceed from one cross section to the next, and unique hydraulic parameters are calculated at each cross section. Computer models, such as HEC-2, perform these calculations and are widely used analytical tools.

Manning's equation for mean velocity, V (in feet per second or meters per second), is given as:

$$V = \frac{k}{n} R^{2/3} S^{1/2}$$

where:

k = 1.486 for English units (1 for metric units)

n = Manning's roughness coefficient

R = hydraulic radius (feet or meters)

S = energy slope (water surface slope).

Manning's roughness coefficient may be thought of as an index of the features of channel roughness that contribute to the dissipation of stream energy. **Table 7.2** shows a range of n values for various boundary materials and conditions.

Two methods are presented for estimating Manning's roughness coefficient for natural channels:

- Direct solution of Manning's equation for n.
- Comparison with computed n values for other channels.

Each method has its own limitations and advantages.

Direct Solution for Determining Manning's n

Even slightly nonuniform flow can be difficult to find in natural channels.

Table 7.2: Manning roughness coefficients for various boundaries.
Source: Ven te Chow 1964.

Boundary	Manning Roughness, *n* Coefficient
Smooth concrete	0.012
Ordinary concrete lining	0.013
Vitrified clay	0.015
Shot concrete, untroweled, and earth channels in best condition	0.017
Straight unlined earth canals in good condition	0.020
Rivers and earth canals in fair condition—some growth	0.025
Winding natural streams and canals in poor condition—considerable moss growth	0.035
Mountain streams with rocky beds and rivers with variable sections and some vegetation along banks	0.040-0.050
Alluvial channels, sand bed, no vegetation	
1. Lower regime	
Ripples	0.017-0.028
Dunes	0.018-0.035
2. Washed-out dunes or transition	0.014-0.024
3. Upper regime	
Plane bed	0.011-0.015
Standing waves	0.012-0.016
Antidunes	0.012-0.020

The method of direct solution for Manning's n does not require perfectly uniform flow. Manning n values are computed for a reach in which multiple cross sections, water surface elevations, and at least one discharge have been measured. A series of water surface profiles are then computed with different n values, and the computed profile that matches the measured profile is deemed to have an n value that most nearly represents the roughness of that stream reach at the specific discharge.

Using Manning's n Measured at Other Channels

The second method for estimating n values involves comparing the reach to a similar reach for which Manning's n has already been computed. This procedure is probably the quickest and most commonly used for estimating Manning's n. It usually involves using values from a table or comparing the study reach with photographs of natural channels. Tables of Manning's n values for a variety of natural and artificial channels are common in the literature on hydrology (Chow 1959, Van Haveren 1986) (Table 7.2). Photographs of stream reaches with computed n values have been compiled by Chow (1959) and Barnes (1967). Estimates should be made for several stages, and the relationship between n and stage should be defined for the range of flows of interest.

Uniform Flow

Under conditions of constant width, depth, area, and velocity, the water surface slope and energy grade line approach the slope of the streambed, producing a condition known as "uniform flow." One feature of uniform flow is that the streamlines are parallel and straight (Roberson and Crowe 1985). Perfectly uniform flow is rarely realized in natural channels, but the condition is approached in some reaches where the geometry of the channel cross section is relatively constant throughout the reach.

Conditions that tend to disrupt uniform flow include bends in the stream course; changes in cross-sectional geometry; obstructions to flow caused by large roughness elements, such as channel bars, large boulders, and woody debris; or other features that cause convergence, divergence, acceleration, or deceleration of flow (**Figure 7.8**). Resistance equations may also be used to evaluate these nonuniform flow conditions (gradually varied flow); however, energy-transition considerations (backwater calculations) must then be factored into the analysis. This requires the use of multiple-transect models (e.g., HEC-2 and WSP2; HEC-2 is a *water surface profile* computer program developed by the U.S. Army Corps of Engineers, Hydrologic Engineering Center, in Davis, California; WSP2 is a similar program developed by the USDA Natural Resources Conservation Service.)

(a) (b) (c)

Figure 7.8: Streamflow paths for channels with constrictions or obstructions.

(a) Riffle or bar, Nisqually, Washington.

Source: J. McShane.

(b) Stream width restriction. (c) Sweeper log.

(d) Stream lines through a reach.

When the roughness coefficient is estimated from table values, the chosen n value (n_b) is considered a base value that may need to be adjusted for additional resistance features. Several publications provide procedures for adjusting base values of n to account for channel irregularities, vegetation, obstructions, and sinuosity (Chow 1959, Benson and Dalrymple 1967, Arcement and Schneider 1984, Parsons and Hudson 1985).

The most common procedure uses the following formula, proposed by Cowan (1959) to estimate the value of n:

$$n = (n_b + n_1 + n_2 + n_3 + n_4)\ m$$

where

- n_b = base value of n for a straight, uniform, smooth channel in natural materials
- n_1 = correction for the effect of surface irregularities
- n_2 = correction for variations in cross section size and shape
- n_3 = correction for obstructions
- n_4 = correction for vegetation and flow conditions
- m = correction for degree of channel meandering

Table 7.3 is taken from Aldridge and Garrett (1973) and may be used to estimate each of the above correction factors to produce a final estimated n.

Energy Equation

The *energy equation* is used to calculate changes in water-surface elevation between two relatively similar cross sections. A simplified version of this equation is:

$$z_1 + d_1 + V_1^2/2g = z_2 + d_2 + V_2^2/2g + h_e$$

where:

- z = minimum elevation of streambed
- d = maximum depth of flow
- V = average velocity
- g = acceleration of gravity
- h_e = energy loss between the two sections

Subscript 1 indicates that the variable is at the upstream cross section, and subscript 2 indicates that the variable is at the downstream cross section.

This simplified equation is applicable when hydraulic conditions between the two cross sections are relatively similar (gradually varied flow) and the channel slope is small (less than 0.18).

Energy losses between the two cross sections occur due to channel boundary roughness and other factors described above. These roughnesses may be represented by a Manning's roughness coefficient, n, and then energy losses can be computed using the Manning equation.

$$h_e = L\ [Qn/kAR^{2/3}]^2$$

where:

- L = distance between cross sections
- Q = discharge
- n = Manning's roughness coefficient
- A = channel cross-sectional area
- R = hydraulic radius (Area/wetted perimeter)
- k = 1 (SI units)
- k = 1.486 (ft-lb-sec units)

Computer models (such as HEC-2 and others) are available to perform these calculations for more complex cross-

Table 7.3: "n" value adjustments.

From Aldridge and Garrett (1973).

	Channel Conditions	n Value Adjustment[1/]	Example
Degree of irregularity (n_1)	Smooth	0.000	Compares to the smoothest channel attainable in a given bed material.
	Minor	0.001-0.005	Compares to carefully dredged channels in good condition but having slightly eroded or scoured side slopes.
	Moderate	0.006-0.010	Compares to dredged channels having moderate to considerable bed roughness and moderately sloughed or eroded side slopes.
	Severe	0.011-0.020	Badly sloughed or scalloped banks of natural streams; badly eroded or sloughed sides of canals or drainage channels; unshaped, jagged, and irregular surfaces of channels in rock.
Variation in channel cross section (n_2)	Gradual	0.000	Size and shape of channel cross sections change gradually.
	Alternating occasionally	0.001-0.005	Large and small cross sections alternate occasionally, or the main flow occasionally shifts from side to side owing to changes in cross-sectional shape.
	Alternating frequently	0.010-0.015	Large and small cross sections alternate frequently, or the main flow frequently shifts from side to side owing to changes in cross-sectional shape.
Effect of obstruction (n_3)	Negligible	0.000-0.004	A few scattered obstructions, which include debris deposits, stumps, exposed roots, logs, piers, or isolated boulders, that occupy less than 5 percent of the cross-sectional area.
	Minor	0.005-0.015	Obstructions occupy less than 15 percent of the cross-sectional area and the spacing between obstructions is such that the sphere of influence around one obstruction does not extend to the sphere of influence around another obstruction. Smaller adjustments are used for curved smooth-surfaced objects than are used for sharp-edged angular objects.
	Appreciable	0.020-0.030	Obstructions occupy from 15 to 20 percent of the cross-sectional area or the space between obstructions is small enough to cause the effects of several obstructions to be additive, thereby blocking an equivalent part of a cross section.
	Severe	0.040-0.050	Obstructions occupy more than 50 percent of the cross-sectional area or the space between obstructions is small enough to cause turbulence across most of the cross section.
Amount of vegetation (n_4)	Small	0.002-0.010	Dense growths of flexible turf grass, such as Bermuda, or weeds growing where the average depth of flow is at least two times the height of the vegetation; supple tree seedlings such as willow, cottonwood, arrowweed, or saltcedar growing where the average depth of flow is at least three times the height of the vegetation.
	Medium	0.010-0.025	Turf grass growing where the average depth of flow is from one to two times the height of the vegetation; moderately dense stemmy grass, weeds, or tree seedlings growing where the average depth of the flow is from two to three times the height of the vegetation; brushy, moderately dense vegetation, similar to 1- to 2-year-old willow trees in the dormant season, growing along the banks and no significant vegetation along the channel bottoms where the hydraulic radius exceeds 2 feet.
	Large	0.025-0.050	Turf grass growing where the average depth of flow is about equal to the height of vegetation; 8- to 10-year-old willow or cottonwood trees intergrown with some weeds and brush (none of the vegetation in foliage) where the hydraulic radius exceeds 2 feet; bushy willows about 1 year old intergrown with some weeds along side slopes (all vegetation in full foliage) and no significant vegetation along channel bottoms where the hydraulic radius is greater than 2 feet.
	Very Large	0.050-0.100	Turf grass growing where the average depth of flow is less than half the height of the vegetation; bushy willow trees about 1 year old intergrown with weeds along side slopes (all vegetation in full foliage) or dense cattails growing along channel bottom; trees intergrown with weeds and brush (all vegetation in full foliage).
Degree of meandering[1/] (adjustment values apply to flow confined in the channel and do not apply where downvalley flow crosses meanders) (m)	Minor	1.00	Ratio of the channel length to valley length is 1.0 to 1.2.
	Appreciable	1.15	Ratio of the channel length to valley length is 1.2 to 1.5.
	Severe	1.30	Ratio of the channel length to valley length is greater than 1.5.

[1/] Adjustments for degree of irregularity, variations in cross section, effect of obstructions, and vegetation are added to the base n value before multiplying by the adjustment for meander.

Manning's n in Relation to Channel Bedforms

Just as Manning's n may vary significantly with changes in stage (water level), channel irregularities, obstructions, vegetation, sinuosity, and bed-material size distribution, n may also vary with bedforms in the channel. The hydraulics of sand and mobile-bed channels produce changes in bedforms as the velocity, stream power, and Froude number increase with discharge. The *Froude number* is a dimensionless number that represents the ratio of inertial forces to gravitational force. As velocity and stream power increase, bedforms evolve from ripples to dunes, to washed-out dunes, to plane bed, to antidunes, to chutes and pools. A stationary plane bed, ripples, and dunes occur when the Froude number (long wave equation) is less than 1 (subcritical flow); washed-out dunes occur at a Froude number equal to 1 (critical flow); and a plane bed in motion, antidunes, and chutes and pools occur at a Froude number greater than 1 (supercritical flow). Manning's n attains maximum values when dune bedforms are present, and minimum values when ripples and plane bedforms are present (Parsons and Hudson 1985).

sectional shapes, including floodplains, and for cases where roughness varies laterally across the cross section (USACE 1991).

Analyzing Composite and Compound Cross Sections

Natural channel cross sections are rarely perfectly uniform, and it may be necessary to analyze hydraulics for very irregular cross sections (compound channel). Streams frequently have overflow channels on one or both sides that carry water only during unusually high flows. Overflow channels and overbank areas, which may also carry out-of-bank flows at various flood stages, usually have hydraulic properties significantly different from those of the main channel. These areas are usually treated as separate subchannels, and the discharge computed for each of these subsections is added to the main channel to compute total discharge. This procedure ignores lateral momentum losses, which could cause n values to be underestimated.

A composite cross section has roughness that varies laterally across the section, but the mean velocity can still be computed by a uniform flow equation without subdividing the section. For example, a stream may have heavily vegetated banks, a coarse cobble bed at its lowest elevations, and a sand bar vegetated with small annual willow sprouts.

A standard hydraulics text or reference (such as Chow 1959, Henderson 1986, USACE 1991, etc.) should be con-

Backwater Effects

Straight channel reaches with perfectly uniform flow are rare in nature and, in most cases, may only be approached to varying degrees. If a reach with constant cross-sectional area and shape is not available, a slightly contracting reach is acceptable, provided there is no significant backwater effect from the constriction. Backwater occurs where the stage vs. discharge relationship is controlled by the geometry downstream of the area of interest (e.g., a high riffle controls conditions in the upstream pool at low flow). Manning's equation assumes uniform flow conditions. Manning's equation used with a single cross section, therefore, will not produce an accurate stage vs. discharge relationship in backwater areas. In addition, expanding reaches also should be avoided since there are additional energy losses associated with channel expansions. When no channel reaches are available that meet or approach the condition of uniform flow, it might be necessary to use multitransect models (e.g., HEC-2) to analyze cross section hydraulics. If there are elevation restrictions corresponding to given flows (e.g., flood control requirements), the water surface profile for the entire reach is needed and use of a multitransect (backwater) model is required.

sulted for methods of computing a composite n value for varying conditions across a section and for varying depths of flow.

Reach Selection

The intended use of the cross section analysis plays a large role in locating the reach and cross sections. Cross sections can be located in either a short critical reach where hydraulic characteristics change or in a reach that is considered representative of some larger area. The reach most sensitive to change or most likely to meet (or fail to meet) some important condition may be considered a critical reach. A representative reach typifies a definable extent of the channel system and is used to describe that portion of the system (Parsons and Hudson 1985).

Once a reach has been selected, the channel cross sections should be measured at locations considered most suitable for meeting the uniform flow requirements of Manning's equation. The uniform flow requirement is approached by siting cross sections where channel width, depth, and cross-sectional flow area remain relatively constant within the reach, and the water surface slope and energy grade line approach the slope of the streambed. For this reason, marked changes in channel geometry and discontinuities in the flow (steps, falls, and hydraulic jumps) should be avoided. Generally, sections should be located where it appears the streamlines are parallel to the bank and each other within the selected reach. If uniform flow conditions cannot be met and backwater computations are required, defining cross sections located at changes in channel geometry is essential.

Field Procedures

The basic information to be collected in the reach selected for analysis is a survey of the channel cross sections and water surface slope, a measurement of bed-material particle size distribution, and a discharge measurement. The U.S. Forest Service has produced an illustrated guide to field techniques for stream channel reference sites (Harrelson et al. 1994) that is a good reference for conducting field surveys.

Survey of Cross Section and Water Surface Slope

The cross section is established perpendicular to the flow line, and the points across the section are surveyed relative to a known or arbitrarily established benchmark elevation. The distance/elevation paired data associated with each point on the section may be obtained by sag tape, rod-and-level survey, hydrographic surveys, or other methods.

Water surface slope is also required for a cross section analysis. The survey of water surface slope is somewhat more

> ***Standard Step Backwater Computation***
>
> Many computer programs (e.g., HEC-2) are available to compute water surface profiles. The standard step method of Chow (1959, p. 265) can be used to determine the water surface elevation (depth) at the upstream end of the reach by iterative approximations. This method uses trial water surface elevations to determine the elevation that satisfies the energy and Manning equations written for the end sections of the reach. In using this method, cross sections should be selected so that velocities increase or decrease continuously throughout the reach (USACE 1991).

complicated than the cross section survey in that the slope of the water surface at the location of the section (e.g., pool, run, or riffle) must be distinguished from the more constant slope of the entire reach. (See Grant et al. 1990 for a detailed discussion on recognition and characteristics of channel units.) Water surface slope in individual channel reaches may vary significantly with changes in stage and discharge.

For this reason, when water surface slopes are surveyed in the field, the low-water slope may be approximated by the change in elevation over the individual channel unit where the cross section is located, approximately 1 to 5 channel widths in length, while the high-water slope is obtained by measuring the change in elevation over a much longer reach of channel, usually at least 15 to 20 channel widths in length.

Bed Material Particle Size Distribution

Computing mean velocity with resistance equations based on relative roughness, such as the ones suggested by Thorne and Zevenbergen (1985), requires an evaluation of the particle size distribution of the bed material of the stream. For streams with no significant channel armor and bed material finer than medium gravel, bed material samplers developed by the Federal Interagency Sedimentation Project (FISP 1986) may be used to obtain a representative sample of the streambed, which is then passed through a set of standard sieves to determine percent by weight of particles of various sizes. The cumulative percent of material finer than a given size may then be determined.

Particle size data are usually reported in terms of d_i, where i represents some nominal percentile of the distribution and d_i represents the particle size, usually expressed in millimeters, at which i percent of the total sample by weight is finer. For example, 84 percent of the total sample would be finer than the d_{84} particle size. For additional guidance on bed material sampling in sand-bed streams, refer to Ashmore et al. (1988).

For estimating velocity in steep mountain rivers with substrate much coarser than the medium-gravel limitation of FISP samplers, a *pebble count*, in which at least 100 bed material particles are manually collected from the streambed and measured, is used to measure surface particle size (Wolman 1954). At each sample point along a cross section, a particle is retrieved from the bed, and the intermediate axis (not the longest or shortest axis) is measured. The measurements are tabulated as to number of particles occurring within predetermined size intervals, and the percentage of the total number in each interval is then determined. Again, the percentage in each interval is accumulated to give a particle size distribution, and the particle size data are reported as described above. Additional guidance for bed material sampling in coarse-bed streams is provided in Yuzyk (1986). If an armor layer or pavement is present, standard techniques may be employed to characterize bed sediments, as described by Hey and Thorne (1986).

Discharge Measurement

If several discharge measurements can be made over a wide range of flows, relationships among stage, discharge, and other hydraulic parameters may be developed directly. If only one discharge measurement is obtained, it likely will occur during low water and will be useful for defining the lower end of the rating table. If two measurements can be made, it is desirable to have a low-water measurement and a high-water measurement to define both ends of the rating table and to establish the relationship between Manning's n and stage. If high water cannot be measured directly, it may be necessary to estimate the high-water n (see the discussion earlier in the chapter).

The Bureau of Reclamation *Water Measurement Manual* (USDI-BOR 1997) is an excellent source of information for measuring channel and stream discharge (**Figure 7.9**). Buchanan and Somers (1969) and Rantz et al. (1982) also provide in-depth discussions of discharge measurement techniques. When equipment is functioning properly and standard procedures are followed correctly, it is possible to measure streamflow to within 5 percent of the true value. The USGS considers a "good" measurement of discharge to account for plus or minus 5 percent and an "excellent" discharge measurement to be within plus or minus 3 percent of the true value.

Figure 7.9: Station measuring discharge. *Permanent stations provide measurements for a wide range of flow, but the necessary measurements can be made in other ways.* **Source: C. Zabawa.**

7.B Geomorphic Processes

In planning a project along a river or stream, awareness of the fundamentals of fluvial geomorphology and channel processes allows the investigator to see the relationship between form and process in the landscape. The detailed study of the fluvial geomorphic processes in a channel system is often referred to as a *geomorphic assessment*. The geomorphic assessment provides the process-based framework to define past and present watershed dynamics, develop integrated solutions, and assess the consequences of restoration activities. A geomorphic assessment generally includes data collection, field investigations, and channel stability assessments. It forms the foundation for analysis and design and is therefore an essential first step in the design process, whether planning the treatment of a single reach or attempting to develop a comprehensive plan for an entire watershed.

Stream Classification

The use of any *stream classification* system is an attempt to simplify what are complex relationships between streams and their watersheds.

Although classification can be used as a communications tool and as part of the overall restoration planning process, the use of a classification system is not required to assess, analyze, and design stream restoration initiatives. The design of a restoration does, however, require site-specific engineering analyses and biological criteria, which are covered in more detail in Chapter 8.

Restoration designs range from simple to complex, depending on whether “no action,” only management techniques, direct manipulation, or combinations of these approaches are used. Complete stream corridor restoration designs require an interdisciplinary approach as discussed in Chapter 4. A poorly designed restoration might be difficult to repair and can lead to more extensive problems.

More recent attempts to develop a comprehensive stream classification system have focused on morphological forms and processes of channels and valley bottoms, and drainage networks. Classification systems might be categorized as systems based on sediment transport processes and systems based on channel response to perturbation.

Stream classification methods are related to fundamental variables and processes that form streams. Streams are classified as either alluvial or non-alluvial. An *alluvial stream* is free to adjust its dimensions, such as width, depth, and slope, in response to changes in watershed sediment discharge. The bed and banks of an alluvial stream are composed of material transported by the river under present flow conditions. Conversely, a *non-alluvial* river, like a bedrock-controlled channel, is not free to adjust. Other conditions, such as a high mountain stream flowing in very coarse glacially deposited materials or streams which are significantly controlled by fallen timber, would suggest a non-alluvial system.

Streams may also be classified as either perennial, intermittent, or ephemeral, as discussed in Chapter 1. A perennial stream is one that has flow at all times. An intermittent stream has the potential for continued flow, but at times the entire flow is absorbed by the bed material. This may be seasonal in nature. An ephemeral stream has flow only following a rainfall event. When carrying flow, intermittent and ephemeral streams both have characteristics very similar to those of perennial streams.

Advantages of Stream Classification Systems

The following are some advantages of stream classification systems:

- Classification systems promote communication among persons trained in different resource disciplines.
- They also enable extrapolation of inventory data collected on a few channels of each stream class to a much larger number of channels over a broader geographical area.
- Classification helps the restoration practitioner consider the landscape context and determine the expected range of variability for parameters related to channel size, shape, and pattern and composition of bed and bank materials.
- Stream classification also enables the practitioner to interpret the channel-forming or dominant processes active at the site, providing a base on which to begin the process of designing restoration.
- Classified reference reaches can be used as the stable or desired form of the restoration.
- A classification system is also very useful in providing an important cross-check to verify if the selected design values for width/depth ratio, sinuosity, etc., are within a reasonable range for the stream type being restored.

Limitations of Stream Classification Systems

All stream classification systems have limitations that are inherent to their approaches, data requirements, and range of applicabilities. They should be used cautiously and only for establishing some of the baseline conditions on which to base initial restoration planning. Standard design techniques should never be replaced by stream classification alone.

Some limitations of classification systems are as follows:

- Determination of bankfull or channel-forming flow depth may be difficult or inaccurate. Field indicators are often subtle or missing and are not valid if the stream is not stable and alluvial.
- The dynamic condition of the stream is not indicated in most classification systems. The knowledge of whether the stream is stable, aggrading, or degrading or is approaching a critical geomorphic threshold is important for a successful restoration initiative.
- River response to a perturbation or restoration action is

normally not determined from the classification system alone.

- Biological health of a stream is usually not directly determined through a stream classification system.
- A classification system alone should not be used for determining the type, location, and purpose of restoration activities. These are determined through the planning steps in Part II and the design process in Chapter 8.

When the results of stream classification will be used for planning or design, the field data collection should be performed or directed by persons with experience and training in hydrology, hydraulics, terrestrial and aquatic ecology, sediment transport, and river mechanics. Field data collected by personnel with only limited formal training may not be reliable, particularly in the field determination of bankfull indicators and the assessment of channel instability trends.

Stream Classification Systems

Stream Order

Designation of *stream order*, using the Strahler (1957) method, described in Chapter 1, is dependent on the scale of maps used to identify first-order streams. It is difficult to make direct comparisons of the morphological characteristics of two river basins obtained from topographic maps of different scales. However, the basic morphological relationships defined by Horton (1945) and Yang (1971) are valid for a given river basin regardless of maps used, as shown in the case study of the Rogue River Basin (Yang and Stall 1971, 1973).

Horton (1945) developed some basic empirical stream morphology relations, i.e., Horton's law of stream order, stream slope, and stream length. These show that the relationships between stream order, average stream length, and slope are straight lines on semilog paper.

Yang (1971) derived his theory of average stream fall based on an analogy with thermodynamic principles. The theory states that the ratio of average fall (change in bed elevation) between any two stream orders in a given river basin is unity. These theoretical results were supported by data from 14 river basins in the United States with an average fall ratio of 0.995. The Rogue River basin data were used by Yang and Stall (1973) to demonstrate the relationships between average stream length, slope, fall, and number of streams.

Stream order is used in the *River Continuum Concept* (Vannote et al. 1980), described in Chapter 1, to distinguish different levels of biological activity. However, stream order is of little help to planners and designers looking for clues to restore hydrologic and geomorphic functions to stream corridors.

Schumm

Other classification schemes combine morphological criteria with dominant modes of sediment transport. Schumm (1977) identified straight, meandering, and braided channels and related both channel pattern and stability to modes of sediment transport (**Figure 7.10**).

Schumm recognized relatively stable straight and meandering channels, with predominantly suspended sediment load and cohesive bank materials. On the other end of the spectrum are relatively unstable braided streams characterized by predominantly bedload sediment transport and wide, sandy channels with noncohesive bank materials. The intermediate condition is generally represented by meandering mixed-load channels.

Montgomery and Buffington

Schumm's classification system primarily applies to alluvial channels; Montgomery and Buffington (1993) have proposed a similar classification system for alluvial, colluvial, and bedrock streams in the Pacific Northwest that addresses channel response to sediment inputs throughout the drainage network. Montgomery and Buffington recognize six classes of alluvial channels—cascade, step-pool, plane-bed, riffle-pool, regime, and braided (**Figure 7.11**).

The stream types are differentiated on the basis of channel response to sediment inputs, with steeper channels (cascade and step-pool) maintaining their morphology while transmitting increased sediment loads, and low-gradient channels (regime and pool-

Figure 7.10: Classification of alluvial channels.
Schumm's classification system relates channel stability to kind of sediment load and channel type.
Source: Schumm, The Fluvial System. © 1977. Reprinted by permission of John Wiley and Sons, Inc.

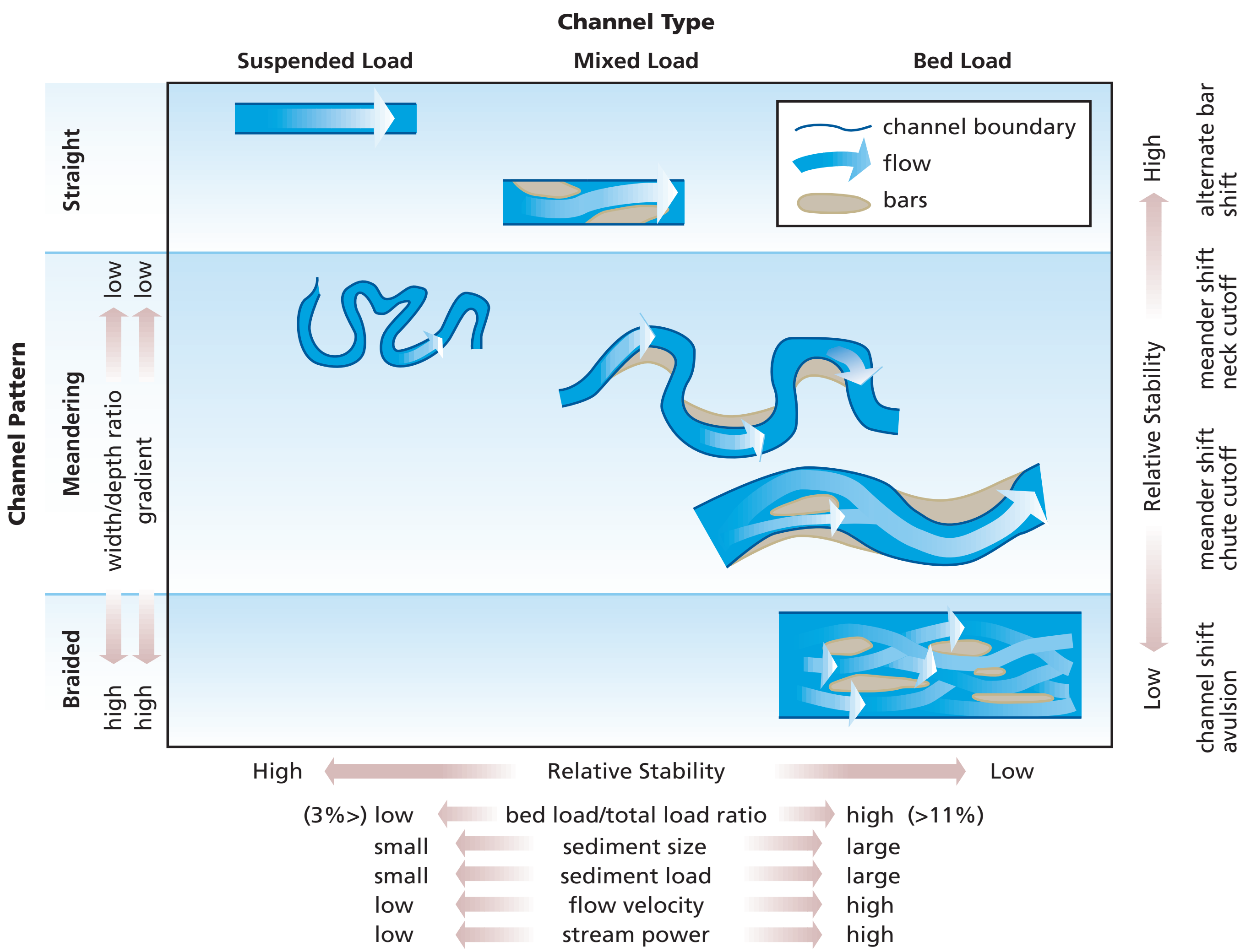
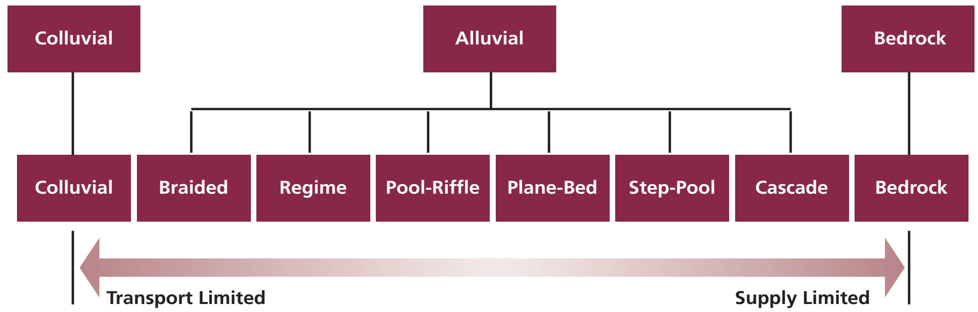


Figure 7.11: Suggested stream classification system for Pacific Northwest.

Included are classifications for nonalluvial streams.

Source: Montgomery and Buffington 1993.



	Braided	Regime	Pool-Riffle	Plane-Bed	Step-Pool	Cascade	Bedrock	Colluvial
Typical Bed Material	Variable	Sand	Gravel	Gravel, cobble	Cobble, boulder	Boulder	N/A	Variable
Bedform Pattern	Laterally oscillary	Multi-layered	Laterally oscillary	None	Vertically oscillary	None	•	Variable
Reach Type	Response	Response	Response	Response	Transport	Transport	Transport	Source
Dominant Roughness Elements	Bedforms (bars, pools)	Sinuosity, bedforms (dunes, ripples, bars) banks	Bedforms (bars, pools), grains, LWD, sinuosity, banks	Grains, banks	Bedforms (steps, pools), grains, LWD, banks	Grains, banks	Boundaries (bed & banks)	Grains, LWD
Dominant Sediment Sources	Fluvial, bank failure, debris flow	Fluvial, bank failure, inactive channel	Fluvial, bank failure, inactive channel, debris flows	Fluvial, bank failure, debris flow	Fluvial, hillslope, debris flow	Fluvial, hillslope, debris flow	Fluvial, hillslope, debris flow	Hillslope, debris flow
Sediment Storage Elements	Overbank, bedforms	Overbank, bedforms, inactive channel	Overbank, bedforms, inactive channel	Overbank, inactive channel	Bedforms	Lee & stoss sides of flow obstructions	•	Bed
Typical Slope (m/m)	$S < 0.03$	$S < 0.001$	$0.001 < S$ and $S < 0.02$	$0.01 < S$ and $S < 0.03$	$0.03 < S$ and $S < 0.08$	$0.08 < S$ and $S < 0.30$	Variable	$S > 0.20$
Typical Confinement	Unconfined	Unconfined	Unconfined	Variable	Confined	Confined	Confined	Confined
Pool Spacing (Channel Widths)	Variable	5 to 7	5 to 7	none	1 to 4	< 1	Variable	Variable

riffle) responding to increased sediment through morphological adjustments. In general, steep channels act as sediment-delivery conduits connecting zones of sediment production with low-gradient response channels.

Rosgen Stream Classification System

One comprehensive stream classification system in common use is based on morphological characteristics described by Rosgen (1996) (**Figure 7.12**). The Rosgen system uses six morphological measurements for classifying a stream reach—entrenchment, width/depth ratio, sinuosity, number of channels, slope, and bedmaterial particle size. These criteria are used to define eight major stream classes with about 100 individual stream types.

Rosgen uses the bankfull discharge to represent the stream-forming discharge or channel-forming flow. Bankfull discharge is needed to use this classification system because all of the morphological relationships are related to this flow condition: width and depth of flow are measured at the bankfull elevation, for example.

Except for entrenchment and width/depth ratio (both of which depend on a determination of bankfull depth), the parameters used are relatively straightforward measurements. The problems in determining bankfull depth were discussed earlier in Chapter 1. The width/depth ratio is taken at bankfull stage and is the ratio of top width to mean depth for the bankfull channel. Sinuosity is the ratio of stream length to valley length or, alternatively, valley slope to stream slope. The bed material particle size used in the classification is the dominant bed surface particle size, determined in the field by a pebble-count procedure (Wolman 1954) or as modified for sand and smaller sizes. Stream slope is measured over a channel reach of at least 20 widths in length.

Entrenchment describes the relationship between a stream and its valley and is defined as the vertical containment of the stream and the degree to which it is incised in the valley floor. It is, therefore, a measure of how accessible a floodplain is to the stream. The entrenchment ratio used in the Rosgen classification system is the flood-prone width of the valley divided by the bankfull width of the channel. Flood-prone width is determined by doubling the maximum depth in the bankfull channel and measuring the width of the valley at that elevation. If the flood-prone width is greater than 2.2 times the bankfull width, the stream is considered to be slightly entrenched or confined and the stream has ready access to its floodplain. A stream is classified as entrenched if its flood-prone width is less than 1.4 times the bankfull width.

A sample worksheet for collecting data and classifying a stream using the Rosgen system is shown in **Figure 7.13**. A field book for collecting reference reach information is available (Leopold et al. 1997).

Channel Evolution Models

Conceptual *models of channel evolution* describe the sequence of changes a stream undergoes after certain kinds of disturbances. The changes can include increases or decreases in the

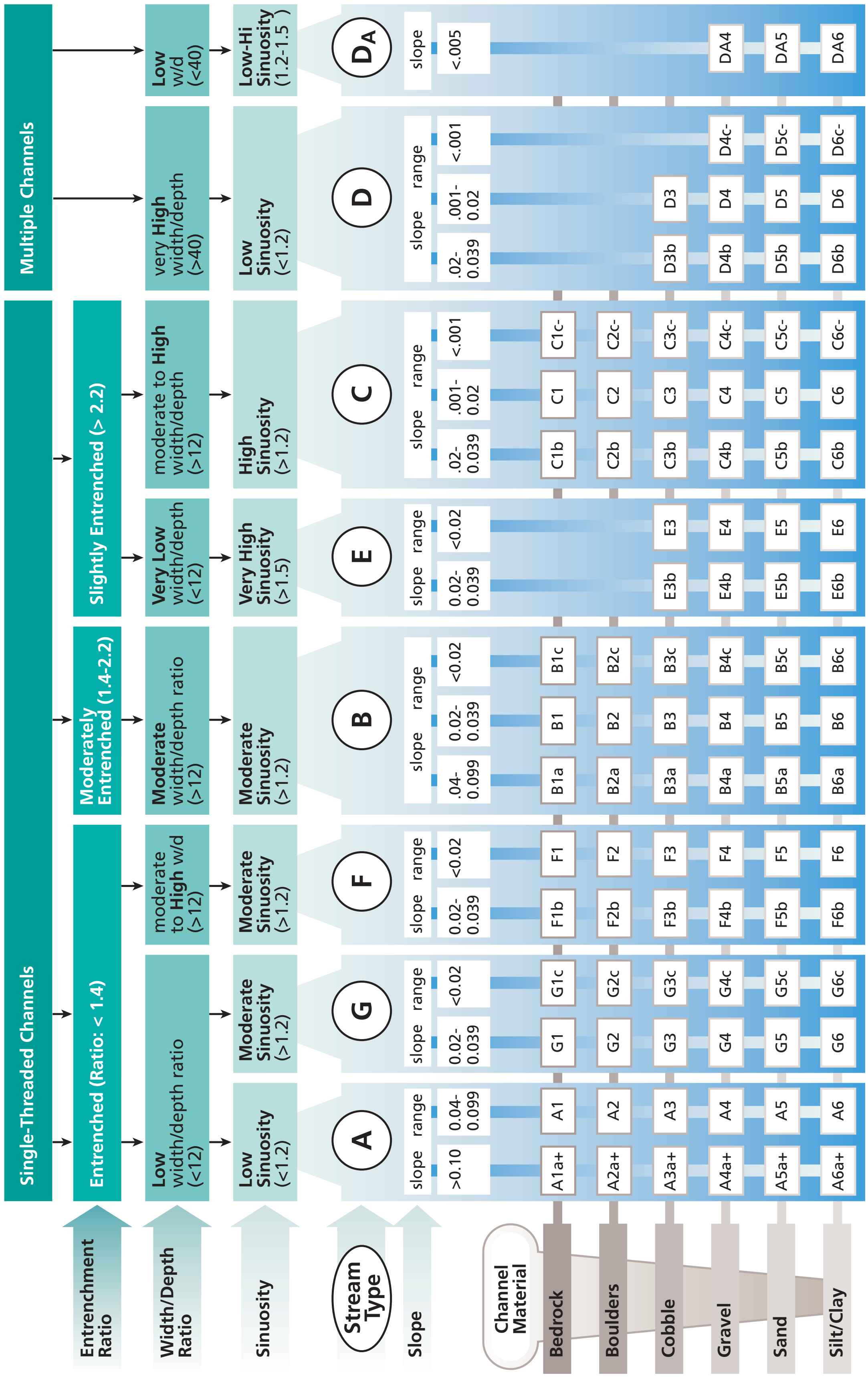


Figure 7.12: Rosgen's stream channel classification system (Level II).

This classification system includes a recognition of specific characteristics of channel morphology and the relationship between the stream and its floodplain.

Source: Rosgen 1996. Published by permission of Wildland Hydrology.

Figure 7.13: Example of stream classification worksheet used with Rosgen methods.

Source: NRCS 1994 (worksheet) and Rosgen 1996 (pebble count). Published by permission of Wildland Hydrology.

STREAM CLASSIFICATION WORKSHEET

Party:______________________ Date:______________________
State:______________________ County:____________________
Stream: ____

Bankfull Measurements: Lat/Long __________________________
Width _______________ Depth ______________ W/D ______________

Sinuosity (Stream Length/Valley Length) or (Valley Slope/Channel Slope):
Strm. Length ___________________ Valley Slope __________________________
Valley Length ___________________ Channel Slope ________________________

Sinuosity $\frac{S_L}{V_L}$ _____________________ Sinuosity $\frac{V_S}{C_S}$ __________________________

Entrenchment Ratio (Floodprone Width/Bankfull Width):
Floodprone width is water level at 2x maximum depth in bankfull cross-section, or width of intermediate floodplain (10-50 yr. event)
Bankfull Width __________________ Floodprone Width ____________________
Entrenchment Ratio ____________________
Slight = 2.2+ Moderate + 1.41-2.2 Entrenched = 1.0-1.4

Dominant Channel Soils:
Bed Material _____________ Left Bank ______________ Right Bank ______________
Description of Soil Profiles (from base of bank to top)
Left: ____
Right: ____

Riparian Vegetation:
Left Bank: _______________________ Right Bank __________________________
% Total Area (Mass) L _________________ R __________________
% Total Ht w/Roots L _________________ R __________________
Ratio of Actual Bank Height to Bankfull Height _____________________________

Bank Slope (Horizontal to Vertical): L ____________________ R _____________________

STREAM TYPE ________________ Remarks ____

PEBBLE COUNT	Site___________________														
Metric (mm)	English (inches)	Particle	Count	Tot #	% Tot	% Cum	Count	Tot #	% Tot	% Cum	Count	Tot #	% Tot	% Cum
<.062	<.002	Silt/Clay												
.062-0.25	.002-.01	Fine Sand												
0.25-.5	.01-.02	Med Sand												
.5-1.0	.02-.04	Coarse Sand												
1.0-2.0	.04-.08	Vy Coarse Sand												
2-8	.08-.32	Fine Gravel												
8-16	.32-.63	Med Gravel												
16-32	.63-1.26	Coarse Gravel												
32-64	1.26-2.51	Vy Coarse Gravel												
64-128	2.51-5.0	Small Cobbles												
128-256	5.0-10.1	Large Cobbles												
256-512	10.1-20.2	Sm Boulders												
512-1024	20.2-40.3	Med Boulders												
1024-2048	40.3-80.6	Lg Boulders												
2048-4096	80.6-161	Vy Lg Boulders												

width/depth ratio of the channel and also involve alterations in the floodplain. The sequence of changes is somewhat predictable, so it is important that the current stage of evolution be identified so appropriate actions can be planned.

Schumm et al. (1984), Harvey and Watson (1986), Simon (1989a) and Simon and Downs (1995) have proposed similar channel evolution models due to bank collapse based on a “space-for-time” substitution, whereby downstream conditions are interpreted as preceding (in time) the immediate location of interest and upstream conditions are interpreted as following (in time) the immediate location of interest. Thus, a reach in the middle of the watershed that previously looked like the channel upstream will evolve to look like the channel downstream.

Downs (1995) reviews a number of classification schemes for interpreting channel processes of lateral and vertical adjustment (i.e., aggradation, degradation, bend migration, and bar formation). When these adjustment processes are placed in a specific order of occurrence, a channel evolution model (CEM) is developed. Although a number of CEMs have been suggested, two models (Schumm et al. 1984 and Simon 1989a, 1995) have gained wide acceptance as being generally applicable for channels with cohesive banks.

Both models begin with a pre-disturbance condition, in which the channel is well vegetated and has frequent interaction with its floodplain. Following a perturbation in the system (e.g., channelization or change in land use), degradation occurs, usually as a result of excess stream power in the disturbed reach. Channel degradation eventually leads to oversteepening of the banks, and when critical bank heights are exceeded, bank failures and mass wasting (the episodic downslope movement of soil and rock) lead to channel widening. As channel widening and mass wasting proceed upstream, an aggradation phase follows in which a new low-flow channel begins to form in the sediment deposits. Upper banks may continue to be unstable at this time. The final stage of evolution is the development of a channel within the deposited alluvium with dimensions and capacity similar to those of the predisturbance channel (Downs 1995). The new channel is usually lower than the predisturbance channel, and the old floodplain now functions primarily as a terrace.

Once streambanks become high, either by downcutting or by sediment deposition on the floodplain, they begin to fail due to a combination of erosion at the base of the banks and mass wasting. The channel continues to widen until flow depths do not reach the depths required to move the sloughed bank materials. Sloughed materials at the base of the banks may begin to be colonized by vegetation. This added roughness helps increase deposition at the base of the banks, and a new small-capacity channel begins to form between the stabilized sediment deposits. The final stage of channel evolution results in a new bankfull channel and active floodplain at a new lower elevation. The original floodplain has been abandoned due to channel incision or excessive sediment deposition and is now termed a terrace.

Schumm et al. (1984) applied the basic concepts of channel evolution to the problem of unstable channelized streams in Mississippi. Simon (1989) built on Schumm's work in a study of channelized streams in Tennessee. Simon's CEM consisted of six stages (**Figure 7.14**). Both models use the cross section, longitudinal profile, and geomorphic processes to distinguish stages of evolution. Both models were developed for landscapes dominated by streams with cohesive banks. However, the same physical processes of evolution can occur in streams with noncohesive banks but not necessarily in the same well-defined stages.

Table 7.4 and **Figure 7.15** show the processes at work in each of Simon's stages.

Advantages of Channel Evolution Models

CEMs are useful in stream corridor restoration in the following ways (Note: Stages are from Simon's 1989 six-stage CEM):

- CEMs help to establish the direction of current trends in disturbed or constructed channels. For example, if a reach of stream is classified as being in Stage IV of evolution (Figure 7.14), more stable reaches should occur downstream and unstable reaches should occur upstream. Once downcutting or incision occurs in a stream (Stage III), the headcut will advance upstream until it reaches a resistant soil layer, the drainage area becomes too small to generate erosive runoff, or the slope flattens to the point that the stream cannot generate enough energy to downcut. Stages IV to VI will follow the headcut upstream.
- CEMs can help to prioritize restoration activities if modification is planned. By stabilizing a reach of stream in early Stage III with grade control measures, the potential degradation of that reach and upstream reaches can be prevented. It also takes less intensive efforts to successfully restore stream reaches in Stages V and VI than to restore those in Stages III and IV.
- CEMs can help match solutions to the problems. Downcutting in Stage III occurs due to the greater capacity of the stream created by construction, or earlier incision, in Stage II. The downcutting in Stage III requires treatments such as grade control aimed at modifying the factors causing the bottom instability. Bank stability problems are dominant in Stages IV and V, so the approaches to stabilization required are different from those for Stage III. Stages I and VI typically require only maintenance activities.
- CEMs can help provide goals or models for restoration. Reaches of streams in Stages I and VI are graded streams, and their profile, form, and pattern can be used as models for restoring unstable reaches.

Figure 7.14: Channel evolution model.

A disturbed or unstable stream is in varying stages of disequilibrium along its length or profile. A channel evolution model theoretically may help predict future upstream or downstream changes in habitat and stream morphology.

From: Simon 1989, USACE 1990.

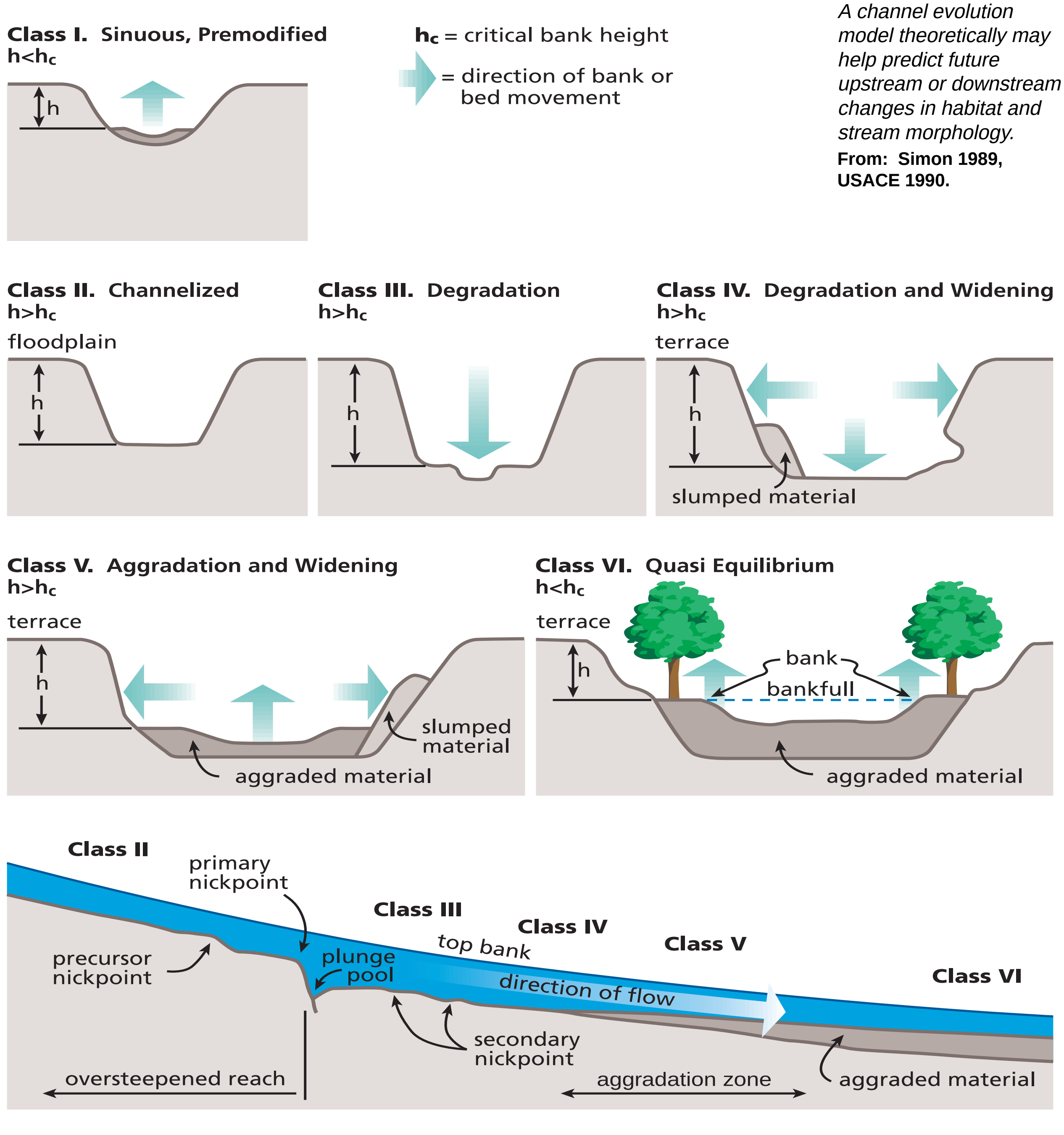


Table 7.4: Dominant hillslope and instream processes, characteristic cross section shape and bedforms, and condition of vegetation in the various stages of channel evolution.
From Simon 1989.

Class		Dominant Processes		Characteristic Forms	Geobotanical Evidence
No.	Name	Fluvial	Hillslope		
I	Premodified	Sediment transport - mild aggradation; basal erosion on outside bends; deposition on inside bends.		Stable, alternate channel bars; convex top-bank shape; flow line high relative to top bank; channel straight or meandering.	Vegetated banks to flow line.
II	Constructed			Trapezoidal cross section; linear bank surfaces; flow line lower relative to top bank.	Removal of vegetation.
III	Degradation	Degradation; basal erosion on banks.	Pop-out failures.	Heightening and steepening of banks; alternate bars eroded; flow line lower relative to top bank.	Riparian vegetation high relative to flow line and may lean towards channel.
IV	Threshold	Degradation; basal erosion on banks.	Slab, rotational and pop-out failures.	Large scallops and bank retreat; vertical face and upper-bank surfaces; failure blocks on upper bank; some reduction in bank angles; flow line very low relative to top bank.	Riparian vegetation high relative to flow line and may lean towards channel.
V	Aggradation	Aggradation; development of meandering thalweg; initial deposition of alternate bars; reworking of failed material on lower banks.	Slab, rotational and pop-out failures; low-angle slides of previously failed material.	Large scallops and bank retreat; vertical face, upper bank, and slough line; flattening of bank angles; flow line low relative to top bank; development of new flood plain.	Tilted and fallen riparian vegetation; reestablishing vegetation on slough line; deposition of material above root collars of slough line vegetation.
VI	Restabilization	Aggradation; further development of meandering thalweg; further deposition of alternate bars; reworking of failed material; some basal erosion on outside bends deposition of flood plain and bank surfaces.	Low-angle slides; some pop-out failures near flow line.	Stable, alternate channel bars; convex-short vertical face on top bank; flattening of bank angles; development of new flood plain; flow line high relative to top bank.	Reestablishing vegetation extends up slough line and upper bank; deposition of material above root collars of slough-line and upper-bank vegetation; some vegetation establishing on bars.

Limitations of Channel Evolution Models

The chief limitations in using CEMs for stream restoration are as follows:

- Future changes in base level elevations and watershed water and sediment yield are not considered when predicting channel response.
- Multiple adjustments by the stream simultaneously are difficult to predict.

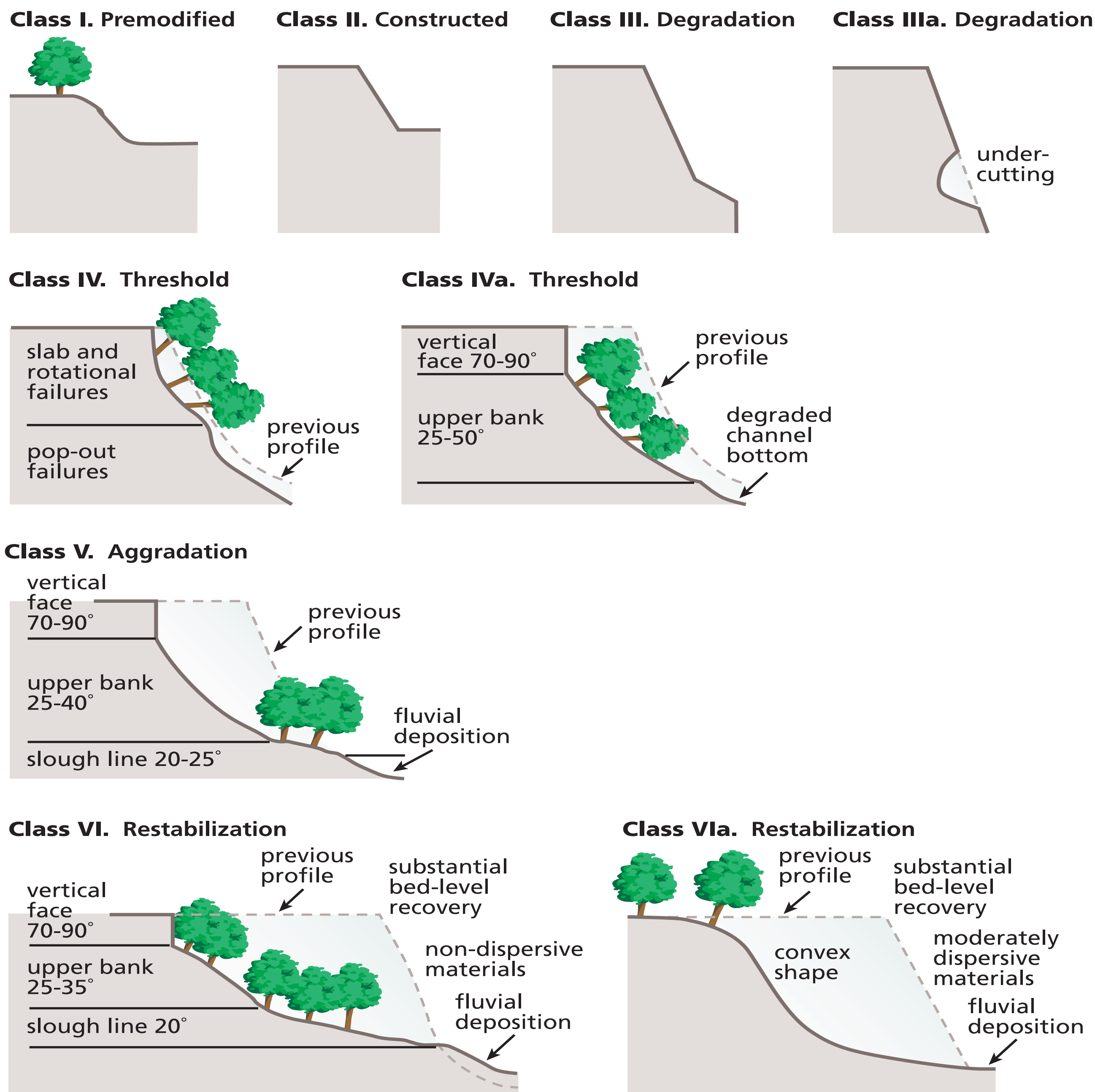
Applications of Geomorphic Analysis

Stream classification systems and channel evolution models may be used together in resource inventories and

Figure 7.15: Simon's channel evolution stages related to streambank shape.

The cross-sectional shape of the streambank may be a good indicator of its evolutionary stage.

Source: Simon 1989. Published by permission of the American Water Resources Association.



analysis to characterize and group streams. Although many classification systems are based on morphological parameters, and channel evolution models are based on adjustment processes, the two approaches to stream characterization complement each other. Both indicate the present condition of a stream reach under investigation, but characterization of additional reaches upstream and downstream of the investigation area can provide an understanding of the overall trend of the stream.

Stream classification systems and channel evolution models also provide insights as to the type of stability problems occurring within the stream corridor and potential opportunities for restoration. Gullied stream channels are downcutting, so grade stabilization is required before time and money are spent on bank stabilization or floodplain restoration. Similarly, incised channels with lateral instabilities are in the initial stages of widening, a process that often must be accommodated before equilibrium conditions can be attained. Although most argue that channel widening must be accommodated to restore incised channels, in some cases not allowing the stream to widen might be preferred, depending on the value and priority placed on adjacent land use and structures within the corridor.

On the other hand, incised streams that have widened enough for a new inner channel and floodplain to begin forming are excellent candidates for vegetation management since these streams are already tending toward renewed stability and establishing riparian vegetation can accelerate the process.

Both the stream classification and the stage of channel evolution inventories can serve as the foundation for assessing systemwide stability. Channel width/depth ratio (F) at mean annual discharge and the percent of silt and clay in the channel boundary (M) are useful diagnostics for determining systemwide adjustments. These variables can be plotted on Schumm's (1960) curve of width/depth ratio versus percent silt-clay ($F = 255M^{-1.08}$) to assess stability (**Figure 7.16**). Schumm's width/depth ratio is the top width of the bankfull channel and the deepest depth in the bankfull channel cross section. The term "M" is defined by the relationship

$$M = [(S_c W) + (S_b 2 D)] / (W + 2 D)$$

where

S_c = percentage of silt and clay in the bed material

S_b = percentage of silt and clay in the bank material

W = channel width

D = channel depth

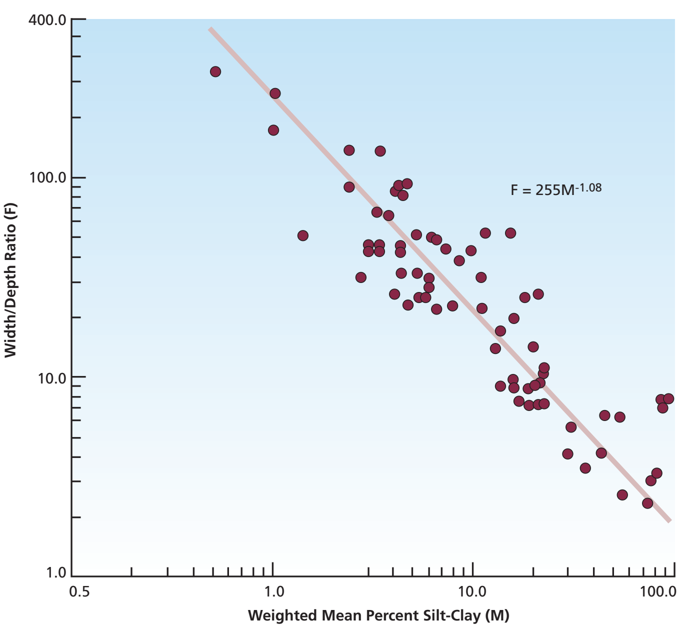


Figure 7.16: Schumm's F versus M relationship. *Data for aggrading streams generally plot above or to the right of the line. Degrading or incising streams plot below the line.* **From Schumm 1960.**

Data from aggrading streams generally plot above the line of best fit, whereas data for degrading streams plot below the line. Schumm's graph could also be used as a guide in selecting an appropriate width/depth ratio for an incised or recently disturbed channel.

Finally, classification systems and evolution models can help guide the selection of restoration treatments. As mentioned above, there is little opportunity for successfully establishing streambank vegetation in streams with vertical and horizontal instability. The banks of such streams are subject to deep-seated slope failures that are not usually prevented even by mature woody vegetation. Conversely, establishing and managing perennial grasses and woody vegetation is critical to protecting streams that are already functioning properly.

Proper Functioning Condition (PFC)

The Bureau of Land Management (BLM) has developed guidelines and procedures to rapidly assess whether a stream riparian area is functioning properly in terms of its hydrology, landform/soils, channel characteristics, and vegetation (Prichard et al. 1993, rev. 1995). This assessment, commonly called PFC, is useful as a baseline analysis of stream condition and physical function, and it can also be useful in watershed analysis.

It is essential to do a thorough analysis of the stream corridor and watershed conditions prior to development of restoration plans and selection of restoration approaches to be used. There are many cases where selection of the wrong approach has led to complete failure of stream restoration efforts and the waste of costs of restoration. In many cases, particularly in wildland situations, restoration through natural processes and control of land uses is the preferred and most cost-effective method. If hydrologic conditions are rapidly changing in a drainage, no restoration might be the wisest course until equilibrium is restored.

Identifying streams and drainages where riparian areas along streams are not in proper functioning condition, and those at risk of losing function, is an important first step in restoration analysis. Physical conditions in riparian zones are excellent indicators of what is happening in a stream or the drainage above.

With the results of PFC analysis, it is possible to begin to determine stream corridor and watershed restoration needs and priorities. PFC results may also be used to identify where gathering more detailed information is needed and where additional data are not needed.

PFC is a methodology for assessing the physical functioning of a riparian-wetland area. It provides information critical to determining the "health" of a riparian ecosystem. PFC considers both abiotic and biotic components as they relate to the physical functioning of riparian areas, but it does not consider the biotic component as it relates to habitat requirements. For habitat analysis, other techniques must be employed.

The PFC procedure is currently a standard baseline assessment for stream/riparian surveys for the BLM, and PFC is beginning to be used by the U.S. Forest Service in the West. This technique is not a substitute for inventory or monitoring protocols designed to yield detailed information on the habitat or populations of plants or animals dependent on the riparian-stream ecosystem.

PFC is a useful tool for watershed analysis. Although the assessment is conducted on a stream reach basis, the ratings can be aggregated and analyzed at the watershed scale. PFC, along with other watershed and habitat condition information, provides a good picture of watershed "health" and causal factors affecting watershed "health." Use of PFC will help to identify watershed-scale problems and suggest management remedies.

The following are definitions of proper function as set forth in TR 1737-9:

- *Proper Functioning Condition*—Riparian-wetland areas are functioning properly when adequate vegetation, landform, or large woody debris is present to:
  1. Dissipate stream energy associated with high waterflows, thereby reducing erosion and improving water quality.
  2. Filter sediment, capture bedload, and aid floodplain development.
  3. Improve floodwater retention and ground water storage.
  4. Develop root masses that stabilize streambanks against cutting action.
  5. Develop diverse ponding and channel characteristics to provide the habitat and the water depth, duration, and temperature necessary for fish production, waterfowl breeding, and other uses.
  6. Support greater biodiversity.
- *Functional-at Risk*—Riparian-wetland areas that are in functional condition, but an existing soil, water, or vegetation attribute makes them susceptible to degradation.
- *Nonfunctional*—Riparian-wetland areas that clearly are not providing adequate vegetation, landform, or large debris to dissipate stream energy associated with high flow and thus are not reducing erosion, improving water quality, or performing other functions as listed above under the definition of proper function. The absence of certain physical attributes, such as absence of a floodplain where one should be, is an indicator of nonfunctioning conditions.

Assessing functionality with the PFC technique involves procedures for determining a riparian-wetland area's capability and potential, and comparing that potential with current conditions.

Although the PFC procedure defines streams without floodplains (when a floodplain would normally be present) as nonfunctional, many streams that lose their floodplains through incision or encroachment still retain ecological functions. The importance of a floodplain needs to be assessed in view of the site-specific aquatic and riparian community.

When using the PFC technique, it is important not to equate “proper function” with “desired condition.” Proper function is intended to describe the state in which the stream channel and associated riparian areas are in a relatively stable and self-sustaining condition. Properly functioning streams can be expected to withstand intermediate flood events (e.g., 25- to 30-year flood events) without substantial damage to existing values. However, proper functioning condition will often develop well before riparian succession provides shrub habitat for nesting birds. Put another way, proper functioning condition is a prerequisite to a variety of desired conditions.

Although based on sound science, the PFC field technique is not quantitative. An advantage of this approach is that it is less time-consuming than other techniques because measurements are not required. The procedure is performed by an interdisciplinary team and involves completing a checklist evaluating 17 factors dealing with hydrology, vegetation, and erosional/depositional characteristics. Training in the technique is required, but the technique is not difficult to learn. With training, the functional determinations resulting from surveys are reproducible to a high degree.

Other advantages of the PFC technique are that it provides an easy-to-understand “language” for discussing stream conditions with a variety of agencies and publics, PFC training is readily available, and there is growing interagency acceptance of the technique.

Hydraulic Geometry: Streams in Cross Section

Stream corridor restoration initiatives frequently involve partial or total reconstruction of channels that have been severely degraded. Channel reconstruction design requires criteria for channel size and alignment. The following material presents an overview of *hydraulic geometry theory* and provides some sample hydraulic geometry relationships for relating bankfull dimensions to bankfull discharge. Correlations between certain planform dimensions (e.g., meander characteristics) of stable alluvial stream channels to bankfull discharge and channel width also are discussed.

Hydraulic geometry theory is based on the concept that a river system tends to develop in a way that produces an approximate equilibrium between the channel and the in-flowing water and sediment (Leopold and Maddock 1953). The theory typically relates an independent or driving variable, such as drainage area or discharge, to dependent variables such as width, depth, slope, and velocity. Hydraulic geometry relations are sometimes stratified according to bed material size or other factors. These relationships are empirically derived, and their development requires a relatively large amount of data.

Figure 7.17: Channel morphology related to average annual discharge
Width, depth, and velocity in relation to mean annual discharge as discharge increases downstream on 19 rivers in Wyoming and Montana.
From Leopold and Maddock 1953.

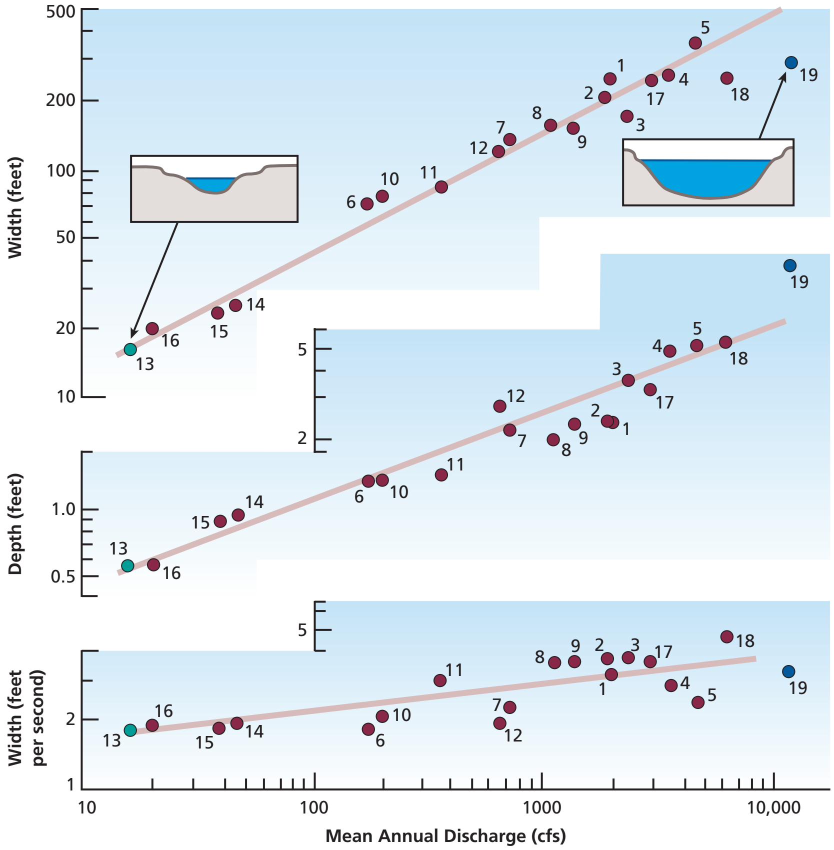


Figure 7.17 presents hydraulic geometry relations based on the mean annual discharge rather than the bankfull discharge. Similar hydraulic geometry relationships can be determined for a watershed of interest by measuring channel parameters at numerous cross sections and plotting them against a discharge. Such plots can be used with care for planning and preliminary design. The use of hydraulic geometry relationships alone for final design is not recommended.

Careful attention to defining stable channel conditions, channel-forming discharge, and streambed and bank characteristics are required in the data collection effort. The primary role of discharge in determining channel cross sections has been clearly demonstrated, but there is a lack of consensus about which secondary factors such as sediment loads, bank materials, and vegetation are significant, particularly with respect to width. Hydraulic geometry relationships that do not explicitly consider sediment transport are applicable mainly to channels with relatively low bed-material loads (USACE 1994).

Hydraulic geometry relations can be developed for a specific river, watershed, or for streams with similar physiographic characteristics. Data scatter is expected about the developed curves even in the same river reach. The more dissimilar the stream and watershed characteristics are, the greater the expected data scatter is. It is important to recognize that this scatter represents a valid range of stable channel configurations due to variables such as geology, vegetation, land use, sediment load and gradation, and runoff characteristics.

Figures 7.18 and 7.19 show hydraulic geometry curves developed for the upper Salmon River watershed in Idaho (Emmett 1975). The scatter of data for stable reaches in the watershed indicates that for a drainage area of 10 square miles, the bankfull discharge could reasonably range from 100 to 250 cfs and the bankfull width could reasonably range from 10 to 35 feet. These relations were developed for a relatively homogeneous watershed, yet there is still quite a bit of

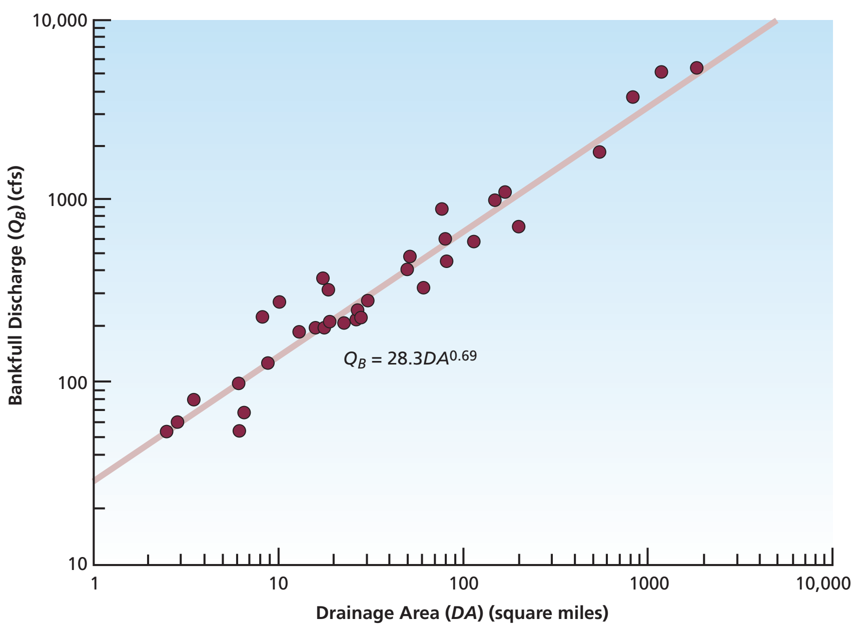


Figure 7.18: Bankfull discharge versus drainage area—Upper Salmon River area.
Curves based on measured data such as this can be valuable tools for designing restorations (Emmett 1975).

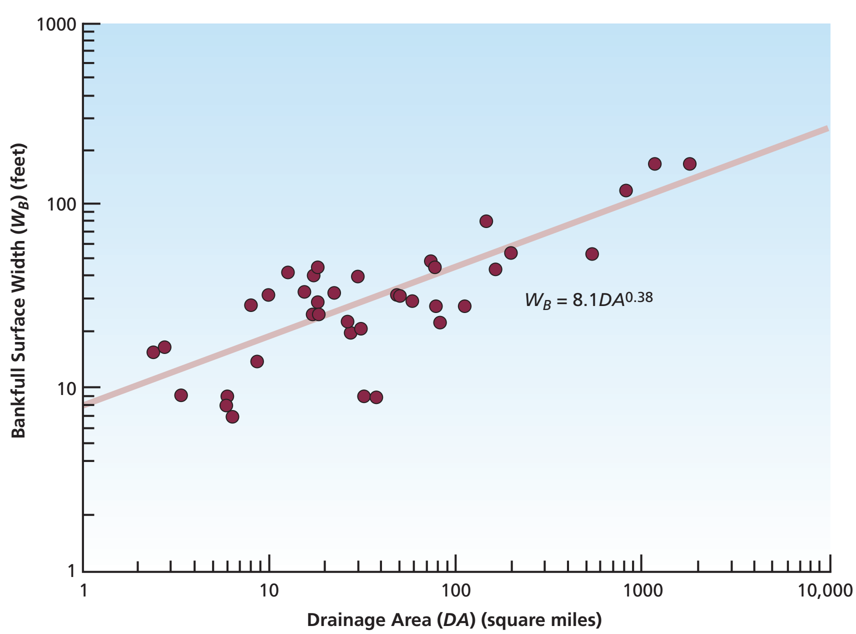


Figure 7.19: Bankfull surface width versus drainage area —Upper Salmon River area.
Local variations in bankfull width may be significant. Road Creek widths are narrower because of lower precipitation.

Regime Theory and Hydraulic Geometry

Regime theory was developed about a century ago by British engineers working on irrigation canals in what is now India and Pakistan. Canals that required little maintenance were said to be "in regime," meaning that they conveyed the imposed water and sediment loads in a state of dynamic equilibrium, with width, depth, and slope varying about some long-term average. These engineers developed empirical formulas linking low-maintenance canal geometry and design discharge by fitting data from relatively straight canals carrying near-constant discharges (Blench 1957, 1969; Simons and Albertson 1963). Since few streams will be restored to look and act as canals, the regime relationships are not presented here.

About 50 years later, hydraulic geometry formulas similar to regime relationships were developed by geomorphologists studying stable, natural rivers. These rivers, of course, were not straight and had varying discharges. A sample of these hydraulic geometry relationship is presented in the table on the following page. In general, these formulas take the form:

$$w = k_1 \; Q^{k_2} \; D_{50}^{\;k_3}$$
$$D = k_4 \; Q^{k_5} \; D_{50}^{\;k_6}$$
$$S = k_7 \; Q^{k_8} \; D_{50}^{\;k_9}$$

where w and D are reach average width and depth in feet, S is the reach average slope, D_{50} is the median bed sediment size in millimeters, and Q is the bankfull discharge in cubic feet per second. These formulas are most reliable for width, less reliable for depth, and least reliable for slope.

natural variation in the data. This illustrates the importance of viewing the data used to develop any curve (not just the curve itself), along with statistical parameters such as R^2 values and confidence limits. (Refer to a text on statistics for additional information.)

Given the natural variation related to stream and watershed characteristics, the preferred source of data for a hydraulic geometry relationship would be the restoration initiative reach. This choice may be untenable due to channel instability. The second preferred choice is the project watershed, although care must be taken to ensure that data are acquired for portions of the watershed with physiographic conditions similar to those of the project reach.

Statistically, channel-forming discharge is a more reliable independent variable for hydraulic geometry relations than drainage area. This is because the magnitude of the channel forming discharge is the driving force that creates the observed channel geometry, and drainage area is merely a surrogate for discharge. Typically, channel-forming discharge correlates best with channel width. Correlations with depth are somewhat less reliable. Correlations with slope and velocity are the least reliable.

Hydraulic Geometry and Stability Assessment

The use of hydraulic geometry relations to assess the stability of a given channel reach requires two things. First, the watershed and stream channel characteristics of the reach in question must be the same as (or similar to) the data set used to develop the hydraulic geometry relations. Second, the reasonable scatter of the data in the hydraulic geometry relations must be known. If the data for a specific reach fall outside the reasonable scatter of data for stable reaches in a similar watershed, there is reason to believe that the reach in question may be unstable. This is only an indicator, since variability in other factors (geology, land use, vegetation, etc.) may cause a given reach to plot high or low on a curve. For instance, in Figure 7.17, the data points from the Road Creek subbasin plot well below the line (narrower bankfull surface width) because the precipitation in this subbasin is lower. These reaches are not unstable; they have developed smaller channel widths in response to lower discharges (as one would expect).

In summary, the use of hydraulic geometry relations requires that the actual data be plotted and the statistical coefficients known. Hydraulic geometry relations can be used as a preliminary guide to indicate stability or instability in stream reaches, but these indications should be checked using other techniques due to the wide natural variability of the data (see Chapter 8 for more information on assessment of channel stability).

Regional Curves

Dunne and Leopold (1978) looked at similar relationships from numerous watersheds and published *regional curves* relating bankfull channel dimensions to drainage area (**Figure 7.20**). Using these curves, the width and depth of the bankfull channel can be approximated once the drainage area of a watershed within one of these regions is known. Obviously, more curves such as these are needed for regions that experience different topographic, geologic, and hydrologic regimes; therefore, additional regional relationships should be developed for specific areas of interest. Several hydraulic geometry formulas are presented in **Table 7.5**.

Regional curves should be used only as indicators to help identify the channel geometry at a restoration initiative site because of the large degree of natural variation in most data sets. Published hydraulic geometry relationships usually are based on stable, single-thread alluvial channels. Channel geometry-discharge relationships are more complex for multithread channels.

Exponents and coefficients for hydraulic geometry formulas are usually determined from data sets for a specific stream or watershed. The relatively small range of variation of the exponents k_2, k_5, and k_8 is impressive, considering the wide range of situations represented. Extremes for the data sets used to generate the hydraulic geometry formulas are given in **Tables 7.6 and 7.7**. Because formula coefficients vary, applying a given set of hydraulic geometry relationships should be limited to channels similar to the calibration sites. This principle severely limits applying the Lacey, Blench, and Simons and Albertson formulas in channel restoration work since these curves were developed using canal data. Additionally, hydraulic geometry relationships developed for pristine or largely undeveloped watersheds should not be applied to urban watersheds.

As shown in Table 7.5, hydraulic geometry relationships for gravel-bed rivers are far more numerous than those for sand-bed rivers. Gravel-bed relationships have been adjusted for bank soil characteristics and vegetation, whereas sand-bed formulas have been modified to include bank silt-clay content (Schumm 1977). Parker (1982) argues in favor of regime-type relationships based on dimensionless variables. Accordingly, the original form of the Parker formula was based on dimensionless variables.

Figure 7.20: Regional curves for bankfull channel dimensions versus drainage area.

Curves showing channel dimensions relating to drainage area for a region of the country can be useful in determining departure from "normal" conditions. The use of such curves must be tempered with an understanding of the limitations of the specific data that produced the curves.

From Dunne and Leopold 1978.

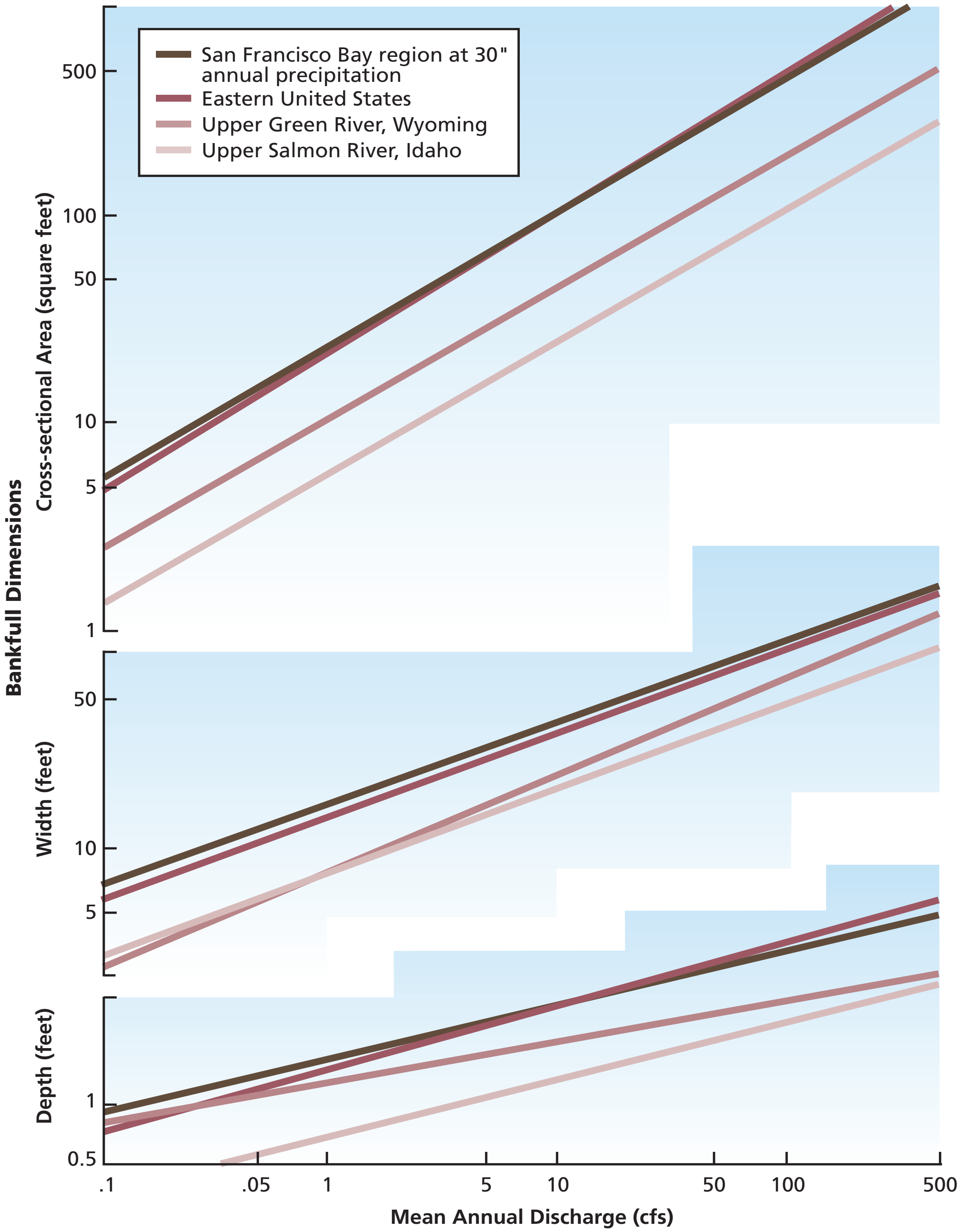


Table 7.5: Limits of data sets used to derive regime formulas.

Source: Hey 1988, 1990.

Reference	Data Source	Median Bed Material Size (mm)	Banks	Discharge (ft^3/s)	Sediment Concentration (ppm)	Slope	Bedforms
Lacey 1958	Indian canals	0.1 to 0.4	Cohesive to slightly cohesive	100 to 10,000	< 500		
Blench 1969	Indian canals	0.1 to 0.6	Cohesive	1 to 100,000	< 30[1]	Not specified	Ripples to dunes
Simons and Albertson 1963	U.S. and Indian canals	0.318 to 0.465	Sand	100 to 400	< 500	.000135 to .000388	Ripples to dunes
		0.06 to 0.46	Cohesive	5 to 88,300	< 500	.000059 to .00034	Ripples to dunes
		Cohesive, 0.029 to 0.36	Cohesive	137 to 510	< 500	.000063 to .000114	Plane
Nixon 1959	U.K. rivers	gravel		700 to 18,050	Not measured		
Kellerhals 1967	U.S., Canadian, and Swiss rivers of low sinuousity, and lab	7 to 265	Noncohesive	1.1 to 70,600	Negligible	.00017 to .0131	Plane
Bray 1982	Sinuous Canadian rivers	1.9 to 145		194 to 138,400	“Mobile” bed	.00022 to .015	
Parker 1982	Single channel Canadian rivers		Little cohesion	353 to 211,900			
Hey and Thorne 1986	Meandering U.K. rivers	14 to 176		138 to 14,970	Q_s computed to range up to 114	.0011 to .021	

[1] Blench (1969) provides adjustment factors for sediment concentrations between 30 and 100 ppm.

Table 7.6: Coefficients for selected hydraulic geometry formulas.

Author	Year	Data	Domain	k_1	k_2	k_3	k_4	k_5	k_6	k_7	k_8	k_9
Nixon	1959	U.K. rivers	Gravel-bed rivers		0.5		0.545	0.33		$1.258n^{2b}$	-0.11	
Leopold et al.	1964	Midwestern U.S.		1.65	0.5			0.4			-0.49	
		Ephemeral streams in semiarid U.S.			0.5			0.3			-0.95	
Kellerhals	1967	Field (U.S., Canada, and Switzerland) and laboratory	Gravel-bed rivers with paved beds and small bed material concentration	1.8	0.5		0.33	0.4	-0.12[a]	0.00062	-0.4	0.92[a]
Schumm	1977	U.S. (Great Plains) and Australia (Riverine Plains of New South Wales)	Sand-bed rivers with properties shown in Table 6	$37k_1*$	0.38		$0.6k_4*$	0.29	-0.12[a]	$0.01136k_7*$	-0.32	
Bray	1982	Canadian rivers	Gravel-bed rivers	3.1	0.53	-0.07	0.304	0.33	-0.03	0.00033	-0.33	0.59
Parker	1982	Single-channel Alberta rivers	Gravel-bed rivers, banks with little cohesion	6.06	0.444	-0.11	0.161	0.401	-0.0025	0.00127	-0.394	0.985
Hay and Thorne	1986	U.K. rivers	Gravel-bed rivers with:									
			Grassy banks with no trees or shrubs	2.39	0.5		0.41	0.37	-0.11	$0.00296k_7**$	-0.43	-0.09
			1-5% tree/ shrub cover	1.84	0.5		0.41	0.37	-0.11	$0.00296k_7**$	-0.43	-0.09
			Greater than 5-50% tree/ shrub cover	1.51	0.5		0.41	0.37	-0.11	$0.00296k_7**$	-0.43	-0.09
			Greater than 50% shrub cover or incised flood plain	1.29	0.5		0.41	0.37	-0.11	$0.00296k_7**$	-0.43	-0.09

[a] Bed material size in Kellerhals' equation is D_{90}.
[b] n = Manning n.
k_1* = $M^{-0.39}$, where M is the percent of bank materials finer than 0.074 mm. The discharge used in this equation is mean annual rather than bankfull.
k_4* = $M^{0.432}$, where M is the percent of bank materials finer than 0.074 mm. The discharge used in this equation is mean annual rather than bankfull.
k_7* = $M^{-0.36}$, where M is the percent of bank materials finer than 0.074 mm. The discharge used in this equation is mean annual rather than bankfull.
k_7** = $D_{54}^{0.84} Q_x^{0.10}$, where Q_x = bed material transport rate in kg s^{-1} at water discharge Q, and D_{54} refers to bed material and is in mm.

Table 7.7: Meander geometry equations.
Source: Williams 1986.

Equation Number	Equation	Applicable Range
Interrelations between meander features		
2	$L_m = 1.25_b$	$18.0 \le L_b \le 43{,}600$ ft
3	$L_m = 1.63B$	$12.1 \le B \le 44{,}900$ ft
4	$L_m = 4.53R_c$	$8.5 \le R_c \le 11{,}800$ ft
5	$L_b = 0.8L_m$	$26 \le L_m \le 54{,}100$ ft
6	$L_b = 1.29B$	$12.1 \le B \le 32{,}800$ ft
7	$L_b = 3.77R_c$	$8.5 \le R_c \le 11{,}800$ ft
8	$B = 0.61L_m$	$26 \le L_m \le 76{,}100$ ft
9	$B = 0.78L_b$	$18.0 \le L_b \le 43{,}600$ ft
10	$B = 2.88R_c$	$8.5 \le R_c \le 11{,}800$ ft
11	$R_c = 0.22L_m$	$33 \le L_m \le 54{,}100$ ft
12	$R_c = 0.26L_c$	$22.3 \le L_b \le 43{,}600$ ft
13	$R_c = 0.35B$	$16 \le B \le 32{,}800$ ft
Relations of channel size to meander features		
14	$A = 0.0094L_m^{1.53}$	$33 \le L_m \le 76{,}100$ ft
15	$A = 0.0149L_b^{1.53}$	$20 \le L_b \le 43{,}600$ ft
16	$A = 0.021B^{1.53}$	$16 \le B \le 38{,}100$ ft
17	$A = 0.117R_c^{1.53}$	$7 \le R_c \le 11{,}800$ ft
18	$W = 0.019L_m^{0.89}$	$26 \le L_m \le 76{,}100$ ft
19	$W = 0.026L_b^{0.89}$	$16 \le L_b \le 43{,}600$ ft
20	$W = 0.031B^{0.89}$	$10 \le B \le 44{,}900$ ft
21	$W = 0.81R_c^{0.89}$	$8.5 \le R_c \le 11{,}800$ ft
22	$D = 0.040L_m^{0.66}$	$33 \le L_m \le 76{,}100$ ft
23	$D = 0.054L_b^{0.66}$	$23 \le L_b \le 43{,}600$ ft
24	$D = 0.055B^{0.66}$	$16 \le B \le 38{,}100$ ft
25	$D = 0.127R_c^{0.66}$	$8.5 \le R_c \le 11{,}800$ ft

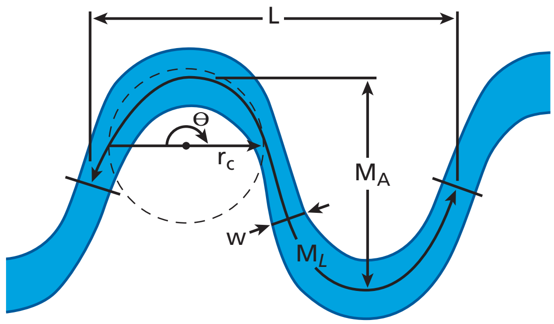
Equation Number	Equation	Applicable Range
Relations of meander features to channel size		
26	$L_m = 21A^{0.65}$	$0.43 \le A \le 225{,}000$ ft
27	$L_b = 15A^{0.65}$	$0.43 \le A \le 225{,}000$ ft
28	$B = 13A^{0.65}$	$0.43 \le A \le 225{,}000$ ft
29	$R_c = 4.1A^{0.65}$	$0.43 \le A \le 225{,}000$ ft
30	$L_m = 6.5W^{1.12}$	$4.9 \le W \le 13{,}000$ ft
31	$L_b = 4.4W^{1.12}$	$4.9 \le W \le 7{,}000$ ft
32	$B = 3.7W^{1.12}$	$4.9 \le W \le 13{,}000$ ft
33	$R_c = 1.3W^{1.12}$	$4.9 \le W \le 7{,}000$ ft
34	$L_m = 129D^{1.52}$	$0.10 \le D \le 59$ ft
35	$L_b = 86D^{1.52}$	$0.10 \le D \le 57.7$ ft
36	$B = 80D^{1.52}$	$0.10 \le D \le 59$ ft
37	$R_c = 23D^{1.52}$	$0.10 \le D \le 57.7$ ft
Relations between channel width, channel depth, and channel sinuosity		
38	$W = 12.5D^{1.45}$	$0.10 \le D \le 59$ ft
39	$D = 0.17W^{0.89}$	$4.92 \le W \le 13{,}000$ ft
40	$W = 73D^{1.23}K^{-2.35}$	$0.10 \le D \le 59$ ft and $1.20 \le K \le 2.60$
41	$D = 0.15w^{0.50}K^{1.48}$	$4.9 \le W \le 13{,}000$ ft and $1.20 \le K \le 2.60$

Derived empirical equations for river-meander and channel-size features.
A = bankfull cross-sectional area.
W = bankfull width.
D = bankfull mean depth.
L_m = meander wavelength.
L_b = along-channel bend length.
B = meander belt width.
R_c = loop radius of curvature.
K = channel sinuosity.

Planform and Meander Geometry: Stream Channel Patterns

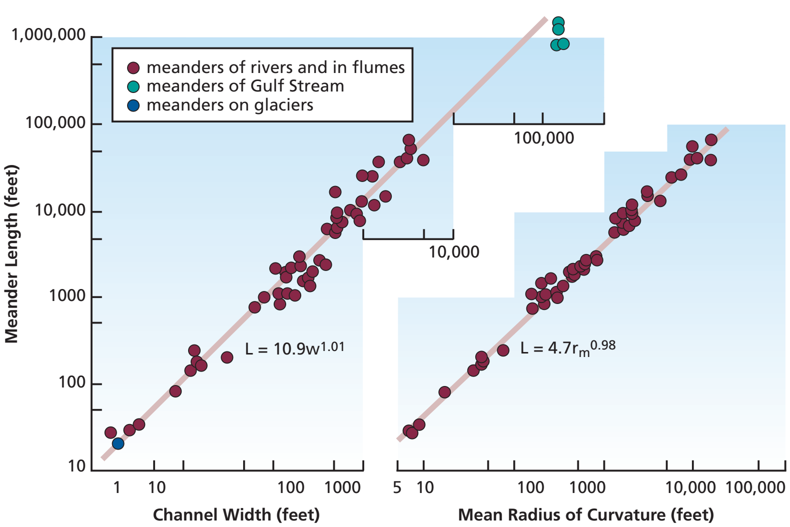
Meander geometry variables are shown in **Figure 7.21**. *Channel planform* parameters may be measured in the field or from aerial photographs and may be compared with published relationships, such as those identified in the box. Developing regional relationships or coefficients specific to the site of interest is, however, preferable to using published relationships that may span wide ranges in value. **Figure 7.22** shows some planform geometry relations by Leopold (1994). Meander geometries that do not fall within the range of predicted relationships may indicate stream instability and deserve attention in restoration design.

Figure 7.21: Meander geometry variables.
Adapted from Williams 1986.



L meander wavelength
M_L meander arc length
w average width at bankfull discharge
M_L meander amplitude
r_c radius of curvature
Θ arc angle

Figure 7.22: Planform geometry relationships.
Meander geometries that do not plot close to the predicted relationship may indicate stream instability.
Source: Leopold 1994.



Stream System Dynamics

Stream management and restoration require knowledge of the complex interactions between watershed and stream processes, boundary sediments, and bank and floodplain vegetation. Identifying the causes of channel instability or potential instability and having knowledge of the magnitude and distribution of channel adjustment processes are important for the following:

- Estimating future channel changes.
- Developing appropriate mitigation measures.
- Protecting the stream corridor.

Adjustment processes that affect entire fluvial systems often include channel incision (lowering of the channel bed with time), aggradation (raising of the channel bed with time), planform geometry changes, channel widening or narrowing, and changes in the magnitude and type of sediment loads. These processes differ from localized processes, such as scour and fill, which can be limited in magnitude and extent.

Meander Geometry Formulas

Reviews of meander geometry formulas are provided by Nunnally and Shields (1985, Table 3) and Chitale (1973). Ackers and Charlton (1970) developed a typical formula that relates meander wavelength and bankfull discharge, Q (cfs), using laboratory data and checking against field data from a wide range of stream sizes:

$$L = 38Q^{0.467}$$

There is considerable scatter about this regression line; examination of the plotted data is recommended. Other formulas, such as this one by Schumm (1977), also incorporate bed sediment size or the fraction of silt-clay in the channel perimeter:

$$L = 1890Q_m^{0.34} / M^{0.74}$$

where Q_m is average discharge (cfs) and M is the percentage of silt-clay in the perimeter of the channel. These types of relationships are most powerful when developed from regional data sets with conditions that are typical of the area being restored. Radius of curvature, r_c, is generally between 1.5 and 4.5 times the channel width, w, and more commonly between 2w and 3w, while meander amplitude is 0.5 to 1.5 times the meander wavelength, L (USACE 1994). Empirical (Apmann 1972, Nanson and Hickin 1983) and analytical (Begin 1981) results indicate that lateral migration rates are greatest for bends with radii of curvature between 2w and 4w.

In contrast, the processes of channel incision and aggradation can affect long reaches of a stream or whole stream systems. Long-term adjustment processes, such as incision, aggradation, and channel widening, can exacerbate local scour problems. Whether streambed erosion occurs due to local scour or channel incision, sufficient bed level lowering can lead to bank instability and to changes in channel planform.

It is often difficult to differentiate between local and systemwide processes without extending the investigation upstream and downstream of the site in question. This is because channels migrate over time and space and so may affect previously undisturbed reaches. For example, erosion at a logjam initially may be attributed to the deflection of flows caused by the woody debris blocking the channel. However, the appearance of large amounts of woody debris may indicate upstream channel degradation related to instability of larger scope.

Determining Stream Instability: Is It Local or Systemwide?

Stage of channel evolution is the primary diagnostic variable for differentiating between local and systemwide channel stability problems in a disturbed stream or constructed channel. During basinwide adjustments, stage of channel evolution usually varies systematically with distance upstream. Downstream sites might be characterized by aggradation and the waning stages of widening, whereas upstream sites might be characterized (in progressive upstream order) by widening and mild degradation, then degradation, and if the investigation is extended far enough upstream, the stable, predisturbed condition (**Figure 7.23**). This sequence of stages can be used to reveal systemwide instabilities. Stream classification can be applied in a similar manner to natural streams. The sequence of stream types can reveal systemwide instabilities.

Restoration measures often fail, not as the result of inadequate structural design, but rather because of the failure of the designers to incorporate the existing and future channel morphology into the design. For this reason, it is important for the designer to have some general understanding of stream processes to ensure that the selected restoration measures will work in harmony with the existing and future river conditions. This will allow the designer to assess whether the conditions at a particular site are due to local instability processes or are the result of some systemwide instability that may be affecting the entire watershed.

Figure 7.23: Bank instability.
Determining if instability is localized or systemwide is imperative to determine a correct path of action.

Systemwide Instability

The equilibrium of a stream system can be disrupted by various factors. Once this occurs, the stream will attempt to regain equilibrium by making adjustments in the dependent variables. These adjustments in the context of physical processes are generally reflected in aggradation, degradation, or changes in planform characteristics (meander wavelength, sinuosity, etc.). Depending on the magnitude of the change and the basin characteristics (bed and bank materials, hydrology, geologic or man-made controls, sediment sources, etc.), these adjustments can propagate throughout the entire watershed and even into neighboring systems. For this reason, this type of disruption of the equilibrium condition is referred to as system instability. If system instability is occurring or expected to occur, it is imperative that the restoration initiative address these problems before any bank stabilization or instream habitat development is considered.

Local Instability

Local instability refers to erosion and deposition processes that are not symptomatic of a disequilibrium condition in the watershed (i.e., system instability). Perhaps the most common form of local instability is bank erosion along the concave bank in a meander bend that is occurring as part of the natural meander process. Local instability can also occur in isolated locations as the result of channel constriction, flow obstructions (ice, debris, structures, etc.), or geotechnical instability. Local instability problems are amenable to local bank protection. Local instability can also exist in channels where severe system instability exists. In these situations, the local instability problems will probably be accelerated due to the system instability, and a more comprehensive treatment plan will be necessary.

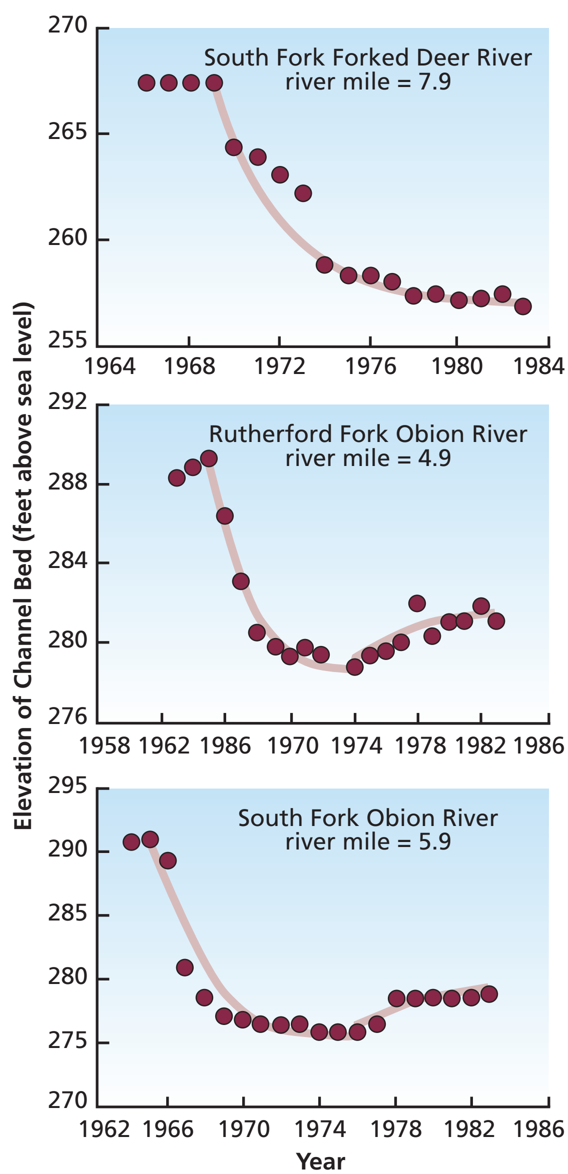
Caution must be exercised if only local treatments on one site are implemented. If the upstream reach is stable and the downstream reach is unstable, a systemwide problem may again be indicated. The instability may continue moving upstream unless the root cause of the instability at the watershed level is removed or channel stabilization at and downstream of the site is implemented.

Local channel instabilities often can be attributed to redirection of flow caused by debris, structures, or the approach angle from upstream. During moderate and high flows, obstructions often result in vortices and secondary-flow cells that accelerate impacts on channel boundaries, causing local bed scour, erosion of bank toes, and ultimately bank failures. A general constriction of the channel cross section from debris accumulation or a bridge causes a backwater condition upstream, with acceleration of the flow and scour through the constriction.

Bed Stability

In unstable channels, the relationship between bed elevation and time (years) can be described by nonlinear functions, where change in response to a disturbance occurs rapidly at first and then slows and becomes asymptotic with time (**Figure 7.24**). Plotting bed elevations against time permits evaluating bed-level adjustment and indicates whether a major phase of channel incision has passed or is ongoing. Various mathematical forms of this function have been used to characterize bed-level adjustment at a site and to predict future bed elevations. This method also can provide valuable information on trends of channel stability at gauged locations where abundant data from discharge measurements are available.

Figure 7.24: Changes in bed elevations over time.
Plotting river bed elevations at a point along the river over time can indicate whether a major phase of channel incision is ongoing or has passed.



Specific Gauge Analysis

Perhaps one of the most useful tools available to the river engineer or geomorphologist for assessing the historical stability of a river system is the specific gauge record. A specific gauge record is a graph of stage for a specific discharge at a particular stream gauging location plotted against time (Blench 1969). A channel is considered to be in equilibrium if the specific gauge record shows no consistent increasing or decreasing trends over time, while an increasing or decreasing trend is indicative of an aggradational or degradational condition, respectively. An example of a specific gauge record is shown in **Figure 7.25.**

The first step in a specific gauge analysis is to establish the stage vs. discharge relationship at the gauge for the period of record being analyzed. A rating curve is developed for each year in the period of record. A regression curve is then fitted to the data and plotted on the scatter plot. Once the rating curves have been developed, the discharges to be used in the specific gauge record must be selected. This selection depends largely on the objectives of the study. It is usually advisable to select discharges that encompass the entire range of observed flows. A plot is then developed showing the stage for the given flow plotted against time.

Specific gauge records are an excellent tool for assessing the historical stability at a specific location. However, specific gauge records indicate only the conditions in the vicinity of the particular gauging station and do not necessarily reflect river response farther upstream or downstream of the gauge. Therefore, even though the specific gauge record is one of the most valuable tools used by river engineers, it should be coupled with other assessment techniques to assess reach conditions or to make predictions about the ultimate response on a river.

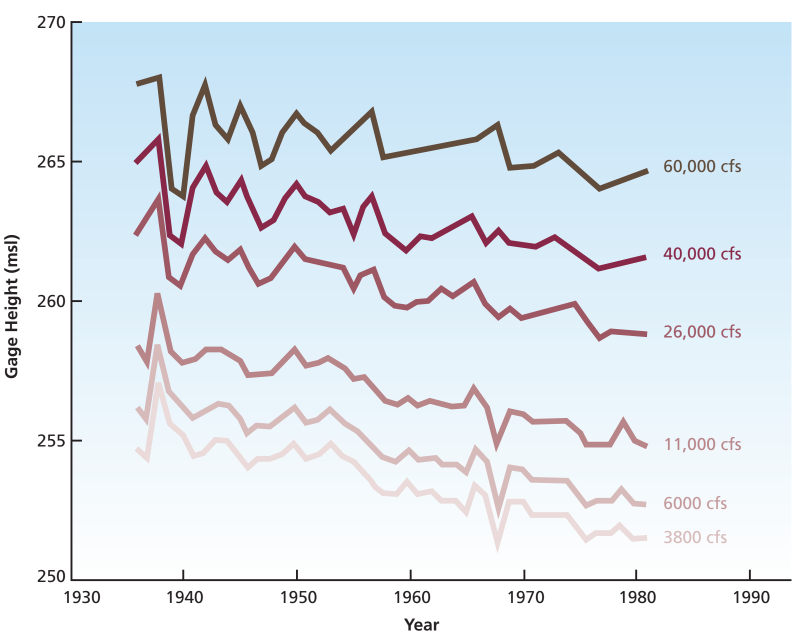


Figure 7.25: Specific gauge plot for Red River at Index, Arkansas.
Select discharges from the gauge data that represent the range of flows.
From USACE 1997 Biedenharn et al.

Comparative Surveys and Mapping

One of the best methods for directly assessing channel changes is to compare channel surveys (thalweg and cross section).

Thalweg surveys are taken along the channel at the lowest point in the cross section. Comparison of several thalweg surveys taken at different points in time allows the engineer or geomorphologist to chart the change in the bed elevation through time (**Figure 7.26**).

Certain limitations that should be considered when comparing surveys on a river system. When comparing thalweg profiles, it is often difficult, especially on larger streams, to determine any distinct trends of aggradation or degradation if there are large scour holes, particularly in bendways. The existence of very deep local scour holes may completely obscure temporal variations in the thalweg. This problem can sometimes be overcome by eliminating the pool sections and focusing only on the crossing locations, thereby allowing aggradational or degradational trends to be more easily observed.

Although thalweg profiles are a useful tool, it must be recognized that they reflect only the behavior of the channel bed and do not provide information about the channel as a whole. For this reason it is usually advisable to study changes in the cross-sectional geometry. Cross-sectional geometry refers to width, depth, area, wetted perimeter, hydraulic radius, and channel conveyance at a specific cross section.

If channel cross sections are surveyed at permanent monumented range locations, the cross-sectional geometry at different times can be compared directly. The cross section plots for each range at the various times can be overlaid and compared. It is seldom the case, however, that the cross sections are located in the exact same place year after year. Because of these problems, it is often advisable to compare reach-average values of the cross-sectional geometry parameters. This requires the study area to be divided into distinct reaches based on geomorphic characteristics. Next, the cross-sectional parameters are calculated at each cross section and then averaged for the entire reach. Then the reach average values can be compared for each survey. Cross-sectional variability between bends (pools) and crossing (riffles) can obscure temporal trends, so it is often preferable to use only cross sections from crossing reaches when analyzing long-term trends of channel change.

Comparison of time-sequential maps can provide insight into the planform instability of the channel. Rates and magnitude of channel migration (bank caving), locations of natural and man-made cutoffs, and spatial and temporal changes in channel width and planform geometry can be determined from maps. With these types of data, channel response to imposed conditions can be documented and used to substantiate predictions of future channel response to a proposed alteration. Planform data can be obtained from aerial photos, maps, or field investigations.

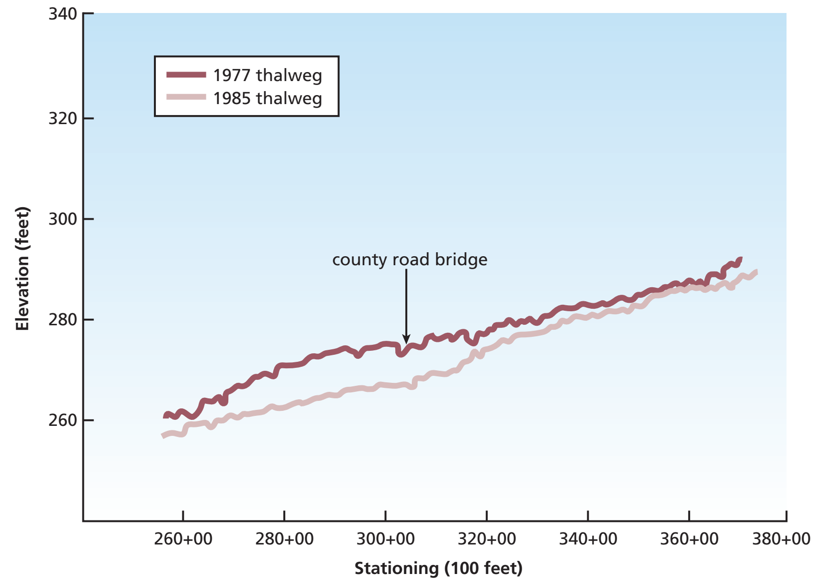


Figure 7.26: Comparative thalweg profiles.
Changes in bed elevation over the length of a stream can indicate areas of transition and reaches where more information is needed.
From USACE 1997 Biedenharn et al.

Regression Functions for Degradation

Two mathematical functions have been used to describe bed level adjustments with time. Both may be used to predict channel response to a disturbance, subject to the caution statements below. The first is a power function (Simon 1989a):

$$E = a\, t^{b}$$

where E = elevation of the channel bed, in feet; a = coefficient, determined by regression, representing the premodified elevation of the channel bed, in feet; t = time since beginning of adjustment process, in years, where t_0 = 1.0 (year prior to onset of the adjustment process); and b = dimensionless exponent, determined by regression and indicative of the non-linear rate of channel bed change (negative for degradation and positive for aggradation).

The second function is a dimensionless form of an exponential equation (Simon 1992):

$$z / z_0 = a + b\, e^{(-k\,t)}$$

where

z = the elevation of the channel bed (at time t)

z_0 = the elevation of the channel bed at t_0

a = the dimensionless coefficient, determined by regression and equal to the dimensionless elevation (z/z_0) when the equation becomes asymptotic, a>1 = aggradation, a<1 = degradation

b = the dimensionless coefficient, determined by regression and equal to the total change in the dimensionless elevation (z/z_0) when the equation becomes asymptotic

k = the coefficient determined by regression, indicative of the rate of change on the channel bed per unit time

t = the time since the year prior to the onset of the adjustment process, in years (t_0=0)

Future elevations of the channel bed can, therefore, be estimated by fitting the equations to bed elevations and by solving for the period of interest. Either equation provides acceptable results, depending on the statistical significance of the fitted relation. Statistical significance of the fitted curves improves with additional data. Degradation and aggradation curves for the same site are fit separately. For degrading sites, the equations will provide projected minimum channel elevations when the value of t becomes large and, by subtracting this result from the floodplain elevation, projected maximum bank heights. A range of bed adjustment trends can be estimated by using different starting dates in the equations when the initial timing of bed level change is unknown. Use of the equations, however, may be limited in some areas because of a lack of survey data.

Regression Functions for Aggradation

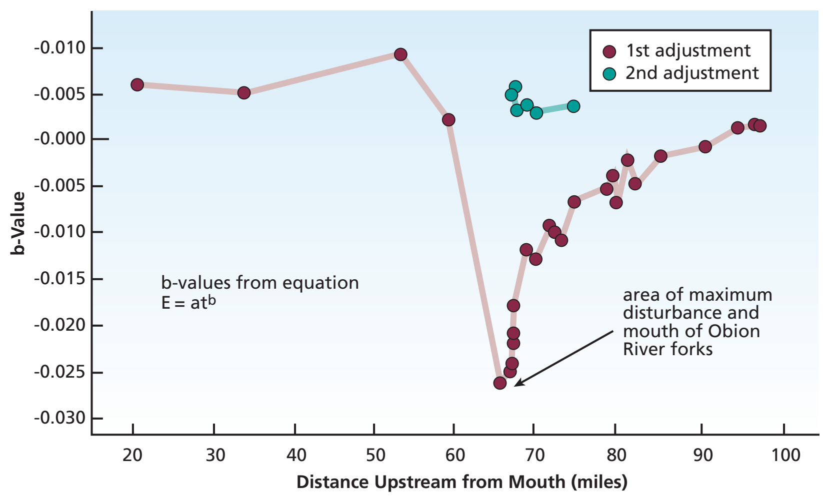
Once the minimum bed elevation has been obtained, that elevation can be used as the starting elevation at a new t_0 for the secondary aggradation phase that occurs during channel widening (see discussion of channel evolution above). Secondary aggradation occurs at a site after degradation reduces channel gradient and stream power to such an extent that sediment loads delivered from degrading reaches upstream can no longer be transported (Simon 1989a). Coefficient values for Simon's power function for estimating secondary aggradation can be obtained either from interpolating existing data or from estimating their values as about 60 percent less than the corresponding value obtained for the degradation phase.

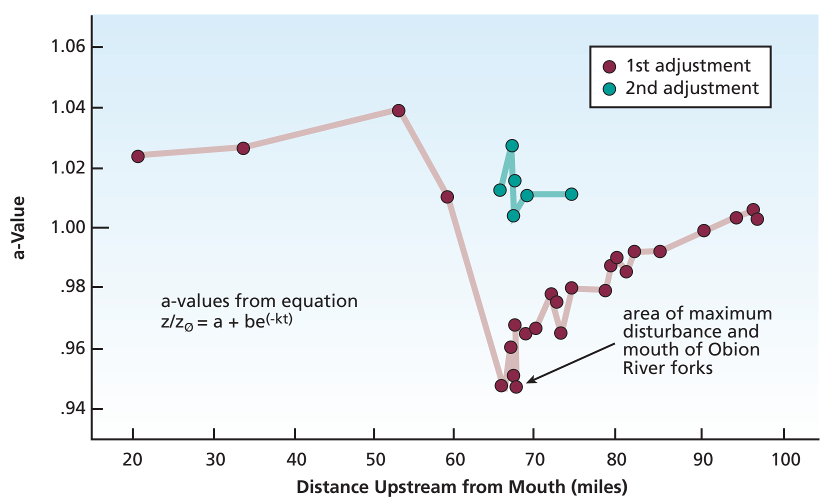
The variation of the regression coefficients a and b with longitudinal distance along the channel can be used as an empirical model of bed level adjustment providing there are data from enough sites. Examples using both equations are provided for the Obion River system, West Tennessee (**Figure 7.27**). Estimates of bed-level change with time for unsurveyed sites can be obtained using interpolated coefficient a values and t_0. For channels downstream from dams without significant tributary sediment inputs, the shape of the a-value curve would be similar but inverted; maximum amounts of degradation (minimum a values) occur immediately downstream of the dam and attenuate nonlinearly with distance farther downstream.

<u>Caution</u>: If one of the above mathematical functions is used to predict future bed elevations, the assumption is made that no new disturbances have occurred to trigger a new phase of channel change. Downstream channelization, construction of a reservoir, formation of a large woody debris jam that blocks the channel, or even a major flood are examples of disturbances that can trigger a new period of rapid change.

The investigator is cautioned that the use of regression functions to compute aggradation and degradation is an empirical approach that might be appropriate for providing insight into the degradational and aggradational processes during the initial planning phases of a project. However, this procedure does not consider the balance between supply and transport of water and sediment and, therefore, is not acceptable for the detailed design of restoration features.

Figure 7.27: Coefficient a and b values for regression functions for estimating bed level adjustment versus longitudinal distance along stream.
Future bed elevations can be estimated by using empirical equations.
From Simon 1989, 1992.





Sediment Transport Processes

This document does not provide comprehensive coverage of sedimentation processes and analyses critical to stream restoration. These processes include erosion, entrainment, transport, deposition, and compaction. Refer to standard texts and reference on sediment, including Vanoni (1975), Simons and Senturk (1977), Chang (1988), Richards (1982), and USACE (1989a).

Numerical Analyses and Models to Predict Aggradation and Degradation

Numerical analyses and models such as HEC-6 are used to predict aggradation and degradation (incision) in stream channels, as discussed in Chapter 8.

Bank Stability

Streambanks can be eroded by moving water removing soil particles or by collapse. Collapse or mass failure occurs when the strength of bank materials is too low to resist gravity forces. Banks that are collapsing or about to collapse are referred to as being geotechnically unstable (**Figure 7.28**). The physical properties of bank materials should be described to aid characterization of potential stability problems and identification of dominant mechanisms of bank instability.

Figure 7.28: Bank erosion by undercutting. *Removal of toe slope support leads to instability requiring geotechnical solutions.*

The level of intensity of geotechnical investigations varies in planning and design. During planning, enough information must be collected to determine the feasibility of alternatives being considered. For example, qualitative descriptions of bank stratigraphy obtained during planning may be all that is required for identifying dominant modes of failure in a study reach. Thorne (1992) describes stream reconnaissance procedures particularly for recording streambank data.

Qualitative Assessment of Bank Stability

Natural streambanks frequently are composed of distinct layers reflecting the depositional history of the bank materials. Each individual sediment layer can have physical properties quite different from those of other layers. The bank profile therefore will respond according to the physical properties of each layer. Since the stability of streambanks with respect to failures due to gravity depends on the geometry of the bank profile and the physical properties of the bank materials, dominant failure mechanisms tend to be closely associated with characteristic stratigraphy or succession of layers (**Figure 7.29**).

A steep bank consisting of uniform layers of cohesive or cemented soils generally develops tension cracks at the top of the bank parallel to the bank alignment. Slab failures occur when the weight of the soil exceeds the strength of the grain-to-grain contacts within the soil. As clay content or cementing agent decreases, the slope of the bank decreases; vertical failure planes become more flat and planar failure surfaces develop. Rotational failures occur when the bank soils are predominantly cohesive. Block-type failures occur when a weak soil layer is eroded away and the layers above the weak layer lose structural support.

The gravity failure processes described in Figure 7.29 usually occur after the banks have been saturated due to precipitation or high stream stages. The water adds weight to the soil and reduces grain-to-grain contacts and cohesion forces while increasing the pore pressure. Pore pressure occurs when soil water in the pore spaces is under pressure from overlying soil and water. Pore pressure therefore is internal to the soil mass. When a stream is full, the flowing water provides some support to the streambanks. When the stream level drops, the internal pore pressure pushes out from within and increases the potential for bank failure.

The last situation described in Figure 7.29 involves ground water sapping or piping. Sapping or piping is the erosion of soil particles beneath the surface by flowing ground water. Dirty or sediment-laden seepage from a streambank indicates ground water sapping or piping is occurring. Soil layers above the areas of ground water piping eventually will collapse after enough soil particles have been removed from the support layer.

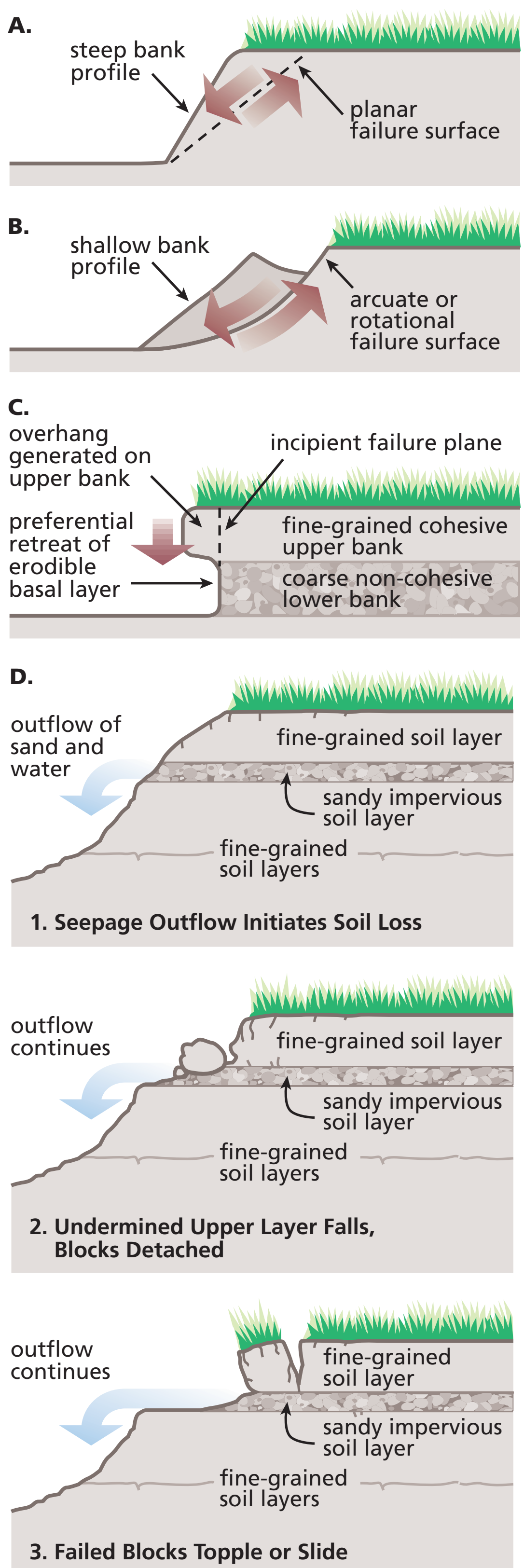
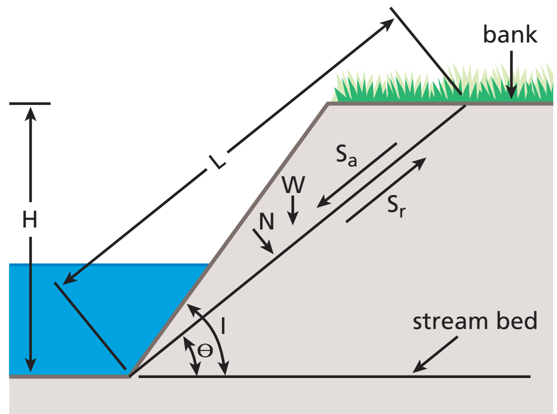


Figure 7.29: Relationship of dominant bank failure mechanisms and associated stratigraphics

(a) Uniform bank undergoing planar type failure (b) Uniform shallow bank undergoing rotational type failure (c) Cohesive upper bank, noncohesive lower bank leads to cantilever type failure mechanism (d) Complex bank stratigraphy may lead to piping or sapping type failures.

From Hagerty 1991. In Journal of Hydraulic Engineering. Vol. 117 Number 8. Reproduced by permission of ASCE.



Explanation

H = bank height
L = failure plane length
c = cohesion
ϕ = friction angle
γ = bulk unit weight
W = weight of failure block
(c, ϕ, γ, W: soil properties)
I = bank angle
S_a = WsinΘ (driving force)
S_r = cL + Ntanϕ (resisting force)
N = WcosΘ
Θ = (0.5I = 0.5ϕ) (failure plane angle)

for the critical case $S_a = S_r$ and:

$$H_c = \frac{4c \sin I \cos \phi}{\gamma\,(1 - \cos [I - \phi])}$$

Figure 7.30: Forces acting on a channel bank assuming there is zero pore-water pressure.
Bank stability analyses relate strength of bank materials to bank height and angles, and to moisture conditions.

Quantitative Assessment of Bank Stability

When restoration design requires more quantitative information on soil properties, additional detailed data need to be collected (**Figure 7.30**). Values of cohesion, friction angle, and unit weight of the bank material need to be quantified. Because of spatial variability, careful sampling and testing programs are required to minimize the amount of data required to correctly characterize the average physical properties of individual layers or to determine a bulk average statistic for an entire bank.

Care must be taken to characterize soil properties not only at the time of measurement but also for the "worst case" conditions at which failure is expected (Thorne et al. 1981). Unit weight, cohesion, and friction angle vary as a function of moisture content. It usually is not possible to directly measure bank materials under worst-case conditions, due to the hazardous nature of unstable sites under such conditions. A qualified geotechnical or soil mechanics engineer should estimate these operational strength parameters.

Quantitative analysis of bank instabilities is considered in terms of force and resistance. The shear strength of the bank material represents the resistance of the boundary to erosion by gravity. Shear strength is composed of cohesive strength and frictional strength. For the case of a planar failure of unit length, the Coulomb equation is applicable

$$S_r = c + (N - \mu) \tan \phi$$

where S_r = shear strength, in pounds per square foot; c = cohesion, in pounds per square foot; N = normal stress, in pounds per square foot; μ = pore pressure, in pounds per square foot; and ϕ = friction angle, in degrees.

Also:

$$N = W \cos \theta$$

where W = weight of the failure block, in pounds per square foot; and θ = angle of the failure plane, in degrees.

The gravitational force acting on the bank is:

$$S_a = W \sin \theta$$

Factors that decrease the erosional resistance (S_r), such as excess pore pressure from saturation and the development of vertical tension cracks, favor bank instabilities. Similarly, increases in bank height (due to channel incision) and bank angle (due to undercutting) favor bank failure by increasing the gravitational force component. In contrast, vegetated banks generally are drier and provide improved bank drainage, which enhances bank stability. Plant roots provide tensile strength to the soil resulting in reinforced earth that resists mass failure, at least to the depth of roots (Yang 1996).

Bank Instability and Channel Widening

Channel widening is often caused by increases in bank height beyond the critical conditions of the bank material. Simon and Hupp (1992) show that there is a positive correlation between the amount of bed level lowering by degradation and amounts of channel widening. The adjustment of channel width by mass-wasting processes represents an important mechanism of channel adjustment and energy dissipation in alluvial streams, occurring at rates covering several orders of magnitude, up to hundreds of feet per year (Simon 1994).

Present and future bank stability may be analyzed using the following procedure:

- Measure the current channel geometry and shear strength of the channel banks.
- Estimate the future channel geometries and model worst-case pore pressure conditions and average shear strength characteristics.

For fine-grained soils, cohesion and friction angle data can be obtained from standard laboratory testing (triaxial shear or unconfined compression tests) or by in situ testing with a borehole shear test device (Handy and Fox 1967, Luttenegger and Hallberg 1981, Thorne et al. 1981, Simon and Hupp 1992). For coarse-grained, cohesionless soils, estimates of friction angles can be obtained from reference manuals. By combining these data with estimates of future bed elevations, relative bank stability can be assessed using bank stability charts.

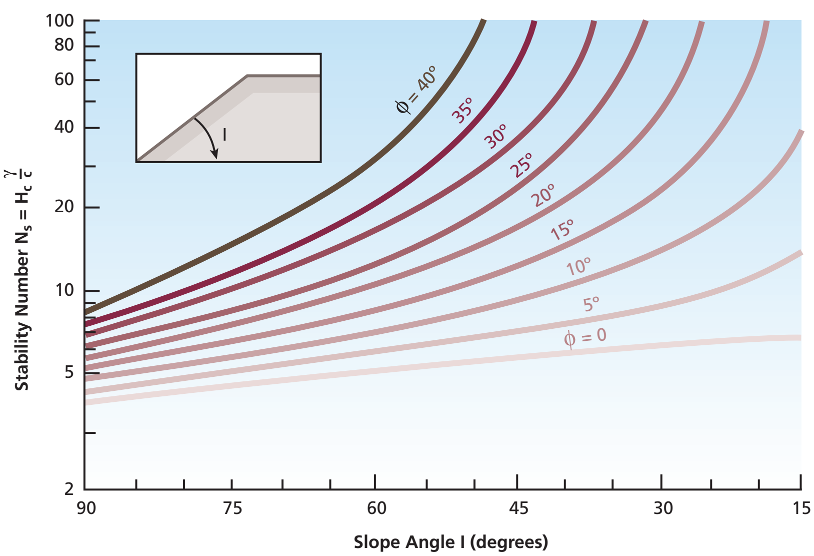
Bank Stability Charts

To produce bank stability charts such as the one following, a stability number (N_s) representing a simplification of the bank (slope) stability equations is used. The stability number is a function of the bank-material friction angle (ϕ) and the bank angle (i) and is obtained from a stability chart developed by Chen (1975) (**Figure 7.31**) or from Lohnes and Handy (1968):

$$N_s = (4 \sin i \cos \phi) / [1 - \cos (i - \phi)]$$

The critical bank height H_c, where driving force S_a = resisting force S_r for a given shear strength and bank geom-

Figure 7.31: Stability number (N_S) as a function of bank angle (i) for a failure surface passing through the bank toe.
Critical bank height for worst case condition can be computed.
From Chen 1975.



etry is then calculated (Carson and Kirkby 1972):

$$H_c = N_s (c / \gamma)$$

where c =cohesion, in pounds per square foot, and γ = bulk unit weight of soil in pounds per cubic foot.

Equations are solved for a range of bank angles using average or ambient soil moisture conditions to produce the upper line "Ambient field conditions, unsaturated." Critical bank height for worst-case conditions (saturated banks and rapid decline in river stage) are obtained by solving the equations, assuming that ϕ and the frictional component of shear strength goes to 0.0 (Lutton 1974) and by using a saturated bulk-unit weight. These results are represented by the lower line, "saturated conditions."

The frequency of b ank failure for the three stability classes (unstable, at-risk, and stable) is subjective and is based primarily on empirical field data (**Figure 7.32**). An unstable channel bank can be expected to fail at least annually and possibly after each major stormflow in which the channel banks are saturated, assuming that there is at least one major stormflow in a given year. At-risk conditions translate to a bank failure every 2 to 5 years, again assuming that there is a major flow event to saturate the banks and to erode toe material. Stable banks by definition do not fail by mass wasting processes. However, channel banks on the outside of meander bends may experience erosion of the bank toe, leading to oversteepening of the bank profile and eventually to bank caving episodes.

Generalizations about critical bank heights (H_c) and angles can be made with knowledge of the variability in cohesive strengths. Five categories of mean cohesive strength of channel banks are identified in **Figure 7.33**. Critical bank heights above the mean low-water level and saturated conditions were used to construct the figure because bank failures typically occur during or after the recession of peak flows. The result is a nomograph giving critical bank heights for a range of bank angles and cohesive strengths

Figure 7.32: Example of a bank stability chart for estimating critical bank height (H_c).

Existing bank stability can be assessed, as well as potential stable design heights and slopes.

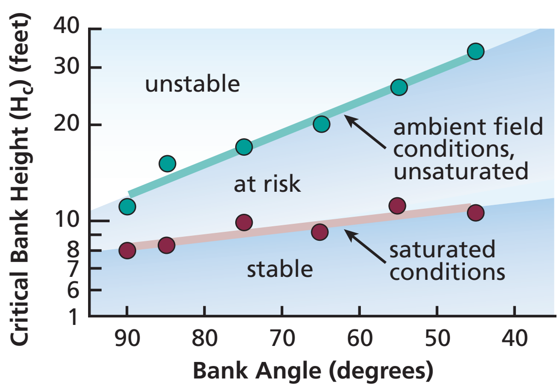
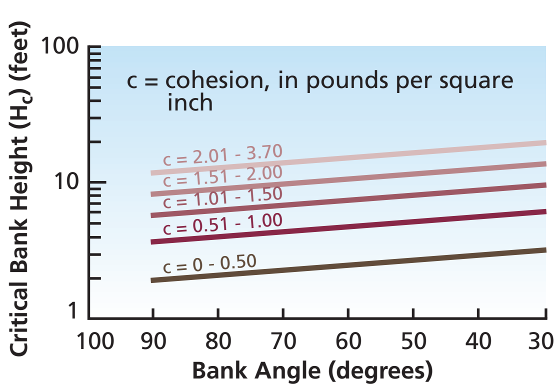


Figure 7.33: Critical bank-slope configurations for various ranges of cohesive strengths under saturated conditions.

Specific data on the cohesive strength of bank materials can be collected to determine stable configurations.



that can be used to estimate stable bank configurations for worst-case conditions, such as saturation during rapid decline in river stage. For example, a saturated bank at an angle of 55 degrees and a cohesive strength of 1.75 pounds per square inch would be unstable when bank heights exceed about 10 feet.

Predictions of Bank Stability and Channel Width

Bank stability charts can be used to determine the following:

- The timing of the initiation of general bank instabilities (in the case of degradation and increasing bank heights).
- The timing of renewed bank stability (in the case of aggradation and decreasing bank heights).
- The bank height and angle needed for a stable bank configuration under a range of moisture conditions.

Estimates of future channel widening also can be made using measured channel-width data over a period of years and then fitting a nonlinear function to the data (**Figure 7.34**). Williams and Wolman (1984) used a dimensionless hyperbolic function of the following form to estimate channel widening downstream from dams:

$$(W_i / W_t) = j_1 + j_2 (1 / t)$$

where:

W_i = initial channel width, in feet

W_t = channel width at t years after W_1, in feet

t = time, in years

j_1 = intercept

j_2 = slope of the fitted straight line on a plot of W_i / W_t versus 1/t

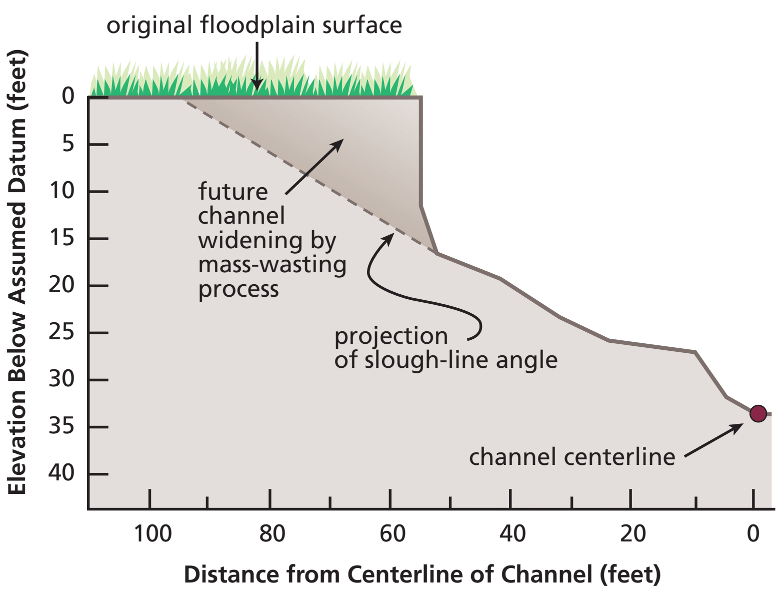


Figure 7.34: Method to estimate future channel widening (10-20 Years) for one side of the channel.

The ultimate bank width can be predicted so that the future stream morphology can be visualized.

Wilson and Turnipseed (1994) used a power function to describe widening after channelization and to estimate future channel widening in the loess area of northern Mississippi:

$$W = x\,t^{d}$$

where:

W = channel width, in feet

x = coefficient, determined by regression, indicative of the initial channel width
t = time, in years
d = coefficient, determined by regression, indicative of the rate of channel widening.

7.C Chemical Characteristics

Assessing water chemistry in a stream restoration initiative can be one of the ways to determine if the restoration was successful. A fundamental understanding of the chemistry of a given system is critical for developing appropriate data collection and analysis methods. Although data collection and analysis are interdependent, each has individual components. It is also critical to have a basic understanding of the hydrologic and water quality processes of interest before data collection and analysis begin. Averett and Schroder (1994) discuss some fundamental concepts used when determining a data collection and analysis program.

Data Collection

Constituent Selection

Hundreds of chemical compounds can be used to describe water quality. It is typically too expensive and too time-consuming to analyze every possible chemical of interest in a given system. In addition to selecting a particular constituent to sample, the analytical techniques used to determine the constituent also must be considered. Another consideration is the chemistry of the constituent; for example, whether the chemical is typically in the dissolved state or sorbed onto sediment makes a profound difference in the methods used for sampling and analysis, as well as the associated costs.

Often it is effective to use parameters that integrate or serve as indicators for a number of other variables. For instance, dissolved oxygen and temperature measurements integrate the net impact of many physical and chemical processes on a stream system, while soluble reactive phosphorus concentration is often taken as a readily available indicator of the potential for growth of attached algae. Averett and Schroder (1993) discuss additional factors involved in selecting constituents to sample.

Sampling Frequency

The needed frequency of sampling depends on both the constituent of interest and management objectives. For instance, a management goal of reducing average instream nutrient concentrations may require monitoring at regular intervals, whereas a goal of maintaining adequate dissolved oxygen (DO) during summer low flow and high temperature periods may require only targeted monitoring during critical conditions. In general, water quality constituents that are highly variable in space or time require more frequent monitoring to be adequately characterized.

In many cases, the concentration of a constituent depends on the flow condition. For example, concentrations of a hydrophobic pesticide, which sorbs strongly to particulate matter, are likely to be highest during scouring flows or erosion washoff events, whereas concentrations of a dissolved chemical that is loaded to the stream at relatively steady rates will exhibit highest concentrations in extremely low flows.

In fact, field sampling and water quality analyses are time-consuming and expensive, and schedule and budget constraints often determine the frequency of data collection. Such constraints make it even more important to design data collection efforts that maximize the value of the information obtained.

Statistical tools often are used to help determine the sampling frequency. Statistical techniques, such as simple random sampling, stratified random sampling, two-stage sampling, and systematic sampling, are described in Gilbert (1987) and Averett and Schroder (1994). Sanders et al. (1983) also describe methods of determining sampling frequency.

Site Selection

The selection of sampling sites is the third critical part of a sampling design. Most samples represent a point in space and provide direct information only on what is happening at that point. A key objective of site selection is to choose a site that gives information that is representative of conditions throughout a particular reach of stream. Because most hydrologic systems are very complex, it is essential to have a fundamental understanding of the area of interest to make this determination.

External inputs, such as tributaries or irrigation return flow, as well as output, such as ground water recharge, can drastically change the water quality along the length of a stream. It is because of these processes that the hydrologic system must be understood to interpret the data from a particular site. For example, downstream from a significant lateral source of a load, the dissolved constituent(s) might be distributed uniformly in the stream channel. Particulate matter, however, typically is stratified. Therefore, the distribution of a constituent sorbed onto particulate matter is not evenly distributed. Averett and Schroder (1994) discuss different approaches to selecting sites to sample both surface water and ground water. Sanders et al. (1983) and Stednick (1991) also discuss site selection.

Finally, practical considerations are an important part of sample collection. Sites first must be accessible, preferably under a full range of potential flow and weather conditions. For this reason, sampling is often conducted at bridge crossings, taking into consideration the degree to which artificial channels at bridge crossings may influence sample results. Finally, where constituent loads and concentrations are of interest, it is important to align water quality sample sites with locations at which flow can be accurately gauged.

Sampling Techniques

This section provides a brief overview of water quality sampling and data collection techniques for stream restoration efforts. Many important issues can be treated only cursorily within the context of this document, but a number of references are available to provide the reader with more detailed guidance.

Key documents describing methods of water sample collection for chemical analysis are the U.S. Geological Survey (USGS) protocol for collecting and processing surface water samples for determining inorganic constituents in filtered water (Horwitz et al. 1994), the field guide for collecting and processing stream water samples for the National Water Quality Assessment program (Shelton 1994), and the field guide for collecting and processing samples of streambed sediment for analyzing trace elements and organic contaminants for the National Water Quality Assessment program (Shelton and Capel 1994). A standard reference document describing methods of sediment collection is the USGS *Techniques for Water-Resource Investigations*, *Field Methods for Measurement of Fluvial Sediment* (Guy and Norman 1982). The USGS is preparing a national field manual that describes techniques for collecting and processing water quality samples (Franceska Wilde, personal communication, 1997).

Sampling Protocols for Water and Sediment

Stream restoration monitoring may involve sampling both water and sediment quality. These samples may be collected by hand (manual samples), by using an automated sampler (automatic samples), as individual point-in-time samples (grab or discrete samples), or combined with other samples (composite samples). Samples collected and mixed in relation to the measured volume within or flow through a system are commonly termed volume- or flow-weighted composite samples, whereas equal-volume samples collected at regular vertical intervals through a portion or all of the water column may be mixed to provide a water column composite sample.

Manual Sampling and Grab Sampling

Samples collected by hand using various types of containers or devices to collect water or sediment from a receiving water or discharge often are termed grab samples. These samples can require little equipment and allow recording miscellaneous additional field observations during each sampling visit.

Manual sampling has several advantages. They approaches are generally uncomplicated and often inexpensive (particularly when labor is already available). Manual sampling is required for sampling some pollutants. For example, according to *Standard Methods* (APHA 1995), oil and grease, volatile compounds, and bacteria must be analyzed from samples collected using manual methods. (Oil, grease, and bacteria can adhere to hoses and jars used in automated sampling equipment, causing inaccurate results; volatile compounds can vaporize during automated sampling procedures or can be lost from poorly sealed sample containers; and bacteria populations can grow and community compositions change during sample storage.)

Disadvantages of grab sampling include the potential for personnel to be available around the clock to sample during storms and the potential for personnel to be exposed to hazardous conditions during sampling.

Long-term sampling programs involving many sampling locations can be expensive in terms of labor costs.

Grab sampling is often used to collect discrete samples that are not combined with other samples. Grab samples can also be used to collect volume- or flow-weighted composite samples, where several discrete samples are combined by proportion to measured volume or flow rates; however, this type of sampling is often more easily accomplished using automated samplers and flow meters. Several examples of manual methods for flow weighting are presented in USEPA (1992a). Grab sampling also may be used to composite vertical water column or aerial composite samples of water or sediment from various kinds of water bodies.

Automatic Sampling

Automated samplers have been improved greatly in the last 10 years and now have features that are useful for many sampling purposes. Generally, such sampling devices require larger initial capital investments or the payment of rental fees, but they can reduce overall labor costs (especially for long-running sampling programs) and increase the reliability of flow-weighted compositing.

Some automatic samplers include an upper part consisting of a microprocessor-based controller, a pump assembly, and a filling mechanism, and a lower part containing a set of glass or plastic sample containers and a well that can be filled with ice to cool the collected samples. More expensive automatic samplers can include refrigeration equipment in place of the ice well; such devices, however, require a 120-volt power supply instead of a battery. Also, many automatic samplers can accept input signals from a flowmeter to activate the sampler and to initiate a flow-weighting compositing program. Some samplers can accept input from a rain gauge to activate a sampling program.

Most automatic samplers allow collecting multiple discrete samples or single or multiple composited samples. Also, samples can be split between sample bottles or can be composited into a single bottle. Samples can be collected on a predetermined time basis or in proportion to flow measurement signals sent to the sampler.

In spite of the obvious advantages of automated samplers, they have some disadvantages and limitations. Some pollutants cannot be sampled by automated equipment unless only qualitative results are desired. Although the cleaning sequence provided by most such samplers provide reasonably separate samples, there is some cross-contamination of the samples since water droplets usually remain in the tubing. Debris in the sampled receiving water can block the sampling line and prevent sample collection. If the sampling line is located in the vicinity of a flowmeter, debris caught on the sampling line can also lead to erroneous flow measurements.

While automatic samplers can reduce manpower needs during storm and runoff events, these devices must be checked for accuracy during these events and must be regularly tested and serviced. If no field checks are made during a storm event, data for the entire event may be lost. Thus, automatic samplers do not eliminate the need for field personnel, but they can reduce these needs and can produce flow-weighted composite samples that might be tedious or impossible using manual methods.

Discrete versus Composite Sampling

Flow rates, physical conditions, and chemical constituents in surface waters often vary continuously and simultaneously. This presents a difficulty when determining water volumes, pollutant concentrations, and masses of pollutants or their loads in the waste discharge flows and in receiving waters. Using automatic or continuously recording flowmeters allows obtaining reasonable and continuous flow rate measurements for these waters. Pollutant loads can then be computed by multiplying these flow volumes over the period of concern by the average pollutant concentration determined from the discrete or flow-composited samples. When manual (instantaneous) flow measurements are used, actual volume flows over time can be estimated only for loading calculations, adding additional uncertainty to loading estimates.

Analyzing constituents of concern in a single grab sample collection provides the minimum information at the minimum cost. Such an approach, however, could be appropriate where conditions are relatively stable; for example, during periods without rainfall or other potential causes of significant runoff and when the stream is well-mixed. Most often, the usual method is to collect a random or regular series of grab samples at predefined intervals during storm or runoff events.

When samples are collected often enough, such that concentration changes between samples are minimized, a clear pattern or time series for the pollutant's concentration dynamics can be obtained. When sampling intervals are spaced too far apart in relation to changes in the pollutant concentration, less clear understanding of these relationships are obtained. Mixing samples from adjacent sampling events or regions (compositing) requires fewer samples to be analyzed; for some assessments, this is a reasonable approach. Sample compositing provides a savings, especially related to costs for water quality analyses, but it also results in loss of information. For example, information on maximum and minimum concentrations during a runoff event is usually lost. But compositing many samples collected through multiple periods during the events can help ensure that the samples analyzed do not include only extreme conditions that are not entirely representative of the event.

Even though analytical results from composited samples rarely equal average conditions for the event, they can still be used, when a sufficient distribution of samples are included, to provide reasonably representative conditions for computing loading estimates. In some analyses, however, considerable errors can be made when using analytical results from composited samples in completing loading analyses. For example, when maximum pollutant concentrations accompany the maximum flow rates, yet concentrations in high and low flows are treated equally, true loadings can be underestimated.

Consequently, when relationships between flow and pollutant concentrations are unknown, it is often preferable initially to include in the monitoring plan at least three discrete or multiple composite sample collections: during the initial period of increasing flow, during the period of the peak or plateau flow, and during the period of declining flow.

The most useful method for sample compositing is to combine samples in relation to the flow volume occurring during study period intervals. There are two variations for accomplishing flow-weighted compositing:

1. Collect samples at equal time intervals at a volume proportional to the flow rate (e.g., collect 100 mL of sample for every 100 gallons of flow that passed during a 10-minute interval) or
2. Collect equal-volume samples at varying times proportional to the flow (e.g., collect a 100-mL sample for each 100 gallons of flow, irrespective of time).

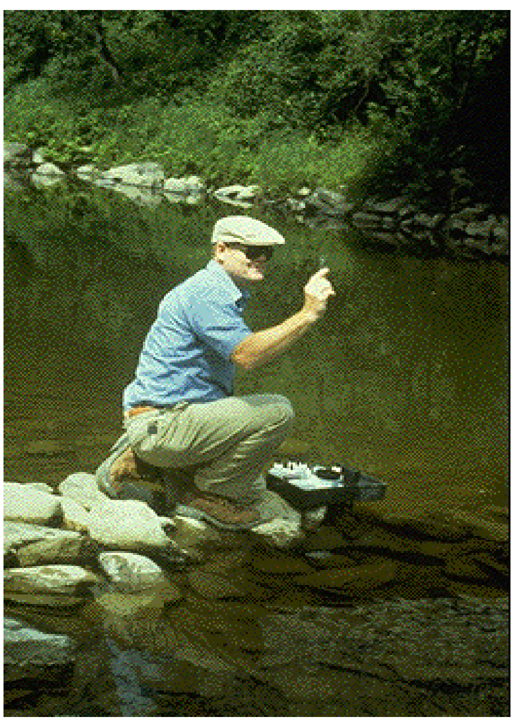


Figure 7.35: Field sampling.
Sampling can also be automated.

The second method is preferable for estimating load accompanying wet weather flows, since it results in samples being collected most often when the flow rate is highest.

Another compositing method is time-composited sampling, where equal sample volumes are collected at equally spaced time intervals (e.g., collect 100 mL of sample every 10 minutes during the monitored event). This approach provides information on the average conditions at the sampling point during the sampling period. It should be used, for example, to determine the average toxic concentrations to which resident aquatic biota are exposed during the monitored event.

Field Analyses of Water Quality Samples

Concentrations of various water quality parameters may be monitored both in the field and in samples submitted to a laboratory (**Figure 7.35**). Some parameters, such as water temperature, must be obtained in the field. Parameters such as concentrations of specific synthetic organic chemicals require laboratory analysis. Other parameters, such as nutrient concentrations, can be measured by both field and laboratory analytical methods. For chemical constituents, field measurements generally should be considered as qualitative screening values since rigorous quality control is not possible. In addition, samples collected for compliance with Clean Water Act requirements must be analyzed by a laboratory certified by the appropriate authority, either the state or the USEPA. The laboratories must use analytic techniques listed in the *Code of Federal Regulations* (CFR), Title 40, Part 136, "Guidelines Establishing Test Procedures for Analysis of Pollutants Under the Clean Water Act."

The balance of this subsection notes special considerations regarding those parameters typically sampled and analyzed in the field, including pH, temperature, and dissolved oxygen (DO).

pH

Levels of pH can change rapidly in samples after collection. Consequently, pH often is measured in the field using a hand-held pH electrode and meter. Electrodes are easily damaged and contaminated and must be calibrated with a standard solution before each use. During calibrations and when site measurements are conducted, field instruments should be at thermal equilibrium with the solutions being measured.

Temperature

Because water temperature changes rapidly after collection, it must be measured either in the field (using in situ probes) or immediately after collecting a grab sample. EPA Method 170.1 describes procedures for thermometric determination of water temperature. Smaller streams often experience wide diurnal variations in temperature, as well as pH and DO. Many streams also experience vertical and longitudinal variability in temperature from shading and flow velocity. Because of the effect of temperature on other water quality factors, such as dissolved oxygen concentration, temperatures always should be recorded when other field measurements are made.

Dissolved Oxygen

When multiple DO readings are required, a DO electrode and meter (EPA method 360.1) are typically used. To obtain accurate measurements, the Winkler titration method should be used to calibrate the meter before and after each day's use. Often it is valuable to recheck the calibration during days of intensive use, particularly when the measurements are of critical importance.

Oxygen electrodes are fragile and subject to contamination, and they need frequent maintenance. Membranes covering these probes must be replaced when bubbles form under the membrane, and the electrode should be kept full of fresh electrolyte solution. If the meter has temperature and salinity compensation controls, they should be used carefully, according to the manufacturer's instructions.

Water Quality Sample Preparation and Handling for Laboratory Analysis

Sample collection, preparation, preservation, and storage guidelines are designed to minimize altering sample constituents. Containers must be made of materials that will not interact with pollutants in the sample, and they should be cleaned in such a way that neither the container nor the cleaning agents interfere with sample analysis. Sometimes, sample constituents must be preserved before they degrade or transform prior to analysis. Also, specified holding times for the sample must not be exceeded. Standard procedures for collecting, preserving, and storing samples are presented in APHA (1995) and at 40 CFR Part 136. Useful material also is contained in the USEPA *NPDES Storm Water Sampling Guidance Document* (1992a).

Most commercial laboratories provide properly cleaned sampling containers with appropriate preservatives. The laboratories also usually indicate the maximum allowed holding periods for each analysis. Acceptable procedures for cleaning sample bottles, preserving their contents, and analyzing for appropriate chemicals are detailed in various methods manuals, including APHA (1995) and USEPA (1979a). Water samplers, sampling hoses, and sample storage bottles always should be made of materials compatible with the goals of the study. For example, when heavy metals are the concern, bottles should not have metal components that can contaminate the collected water samples. Similarly, when organic contaminants are the concern, bottles and caps should be made of materials not likely to leach into the sample.

Sample Preservation, Handling, and Storage

Sample preservation techniques and maximum holding times are presented in APHA (1995) and 40 CFR Part 136. Cooling samples to a temperature of 4 degrees Celsius (°C) is required for most water quality variables. To accomplish this, samples are usually placed in a cooler containing ice or an ice substitute. Many automated samplers have a well next to the sample bottles to hold either ice or ice substitutes. Some more expensive automated samplers have refrigeration equipment requiring a source of electricity. Other preservation techniques include pH adjustment and chemical fixation. When needed, pH adjustments are usually made using strong acids and bases, and extreme care should be exercised when handing these substances.

Bacterial analysis may be warranted, particularly where there are concerns regarding inputs of sewage and other wastes or fecal contamination. Bacterial samples have a short holding time and are not collected by automated sampler. Similarly, volatile compounds must be collected by grab sample, since they are lost through volatilization in automatic sampling equipment.

Sample Labeling

Samples should be labeled with waterproof labels. Enough information should be recorded to ensure that each sample label is unique. The information recorded on sample container labels also should be recorded in a sampling notebook kept by field personnel. The label typically includes the following information:

- Name of project.
- Location of monitoring.
- Specific sample location.
- Date and time of sample collection.
- Name or initials of sampler.

- Analysis to be performed.
- Sample ID number.
- Preservative used.
- Type of sample (grab, composite).

Sample Packaging and Shipping

It is sometimes necessary to ship samples to the laboratory. Holding times should be checked before shipment to ensure that they will not be exceeded. Although wastewater samples are not usually considered hazardous, some samples, such as those with extreme pH, require special procedures. If the sample is shipped through a common carrier or the U.S. Postal Service, it must comply with Department of Transportation Hazardous Material Regulations (49 CFR Parts 171-177). Air shipment of samples defined as hazardous may be covered by the requirements of the International Air Transport Association.

Samples should be sealed in leakproof bags and padded against breakage. Many samples must be packed with an ice substitute to maintain a temperature of 4 degrees C during shipment. Plastic or metal recreational coolers make ideal shipping containers because they protect and insulate the samples. Accompanying paperwork, such as the chain-of-custody documentation, should be sealed in a waterproof bag in the shipping container.

Chain of Custody

Chain-of-custody forms document each change in possession of a sample, starting at its collection and ending when it is analyzed. At each transfer of possession, both the relinquisher and the receiver of the samples are required to sign and date the form. The form and the procedure document possession of the samples and help prevent tampering. The container holding samples also can be sealed with a signed tape or seal to help ensure that samples are not compromised.

Copies of the chain-of-custody form should be retained by the sampler and by the laboratory. Contract laboratories often supply chain-of-custody forms with sample containers. The form is also useful for documenting which analyses will be performed on the samples. These forms typically contain the following information:

- Name of project and sampling locations.
- Date and time that each sample is collected.
- Names of sampling personnel.
- Sample identification names and numbers.
- Types of sample containers.
- Analyses performed on each sample.
- Additional comments on each sample.
- Names of all those transporting the samples.

Collecting and Handling Sediment Quality Samples

Sediments are sinks for a wide variety of materials. Nonpoint source discharges typically include large quantities of suspended material that settle out in sections of receiving waters having low water velocities. Nutrients, metals, and organic compounds can bind to suspended solids and settle to the bottom of a water body when flow velocity is insufficient to keep them in suspension. Contaminants bound to sediments may remain separated from the water column, or they may be resuspended in the water column.

Flood scouring, bioturbation (mixing by biological organisms), desorption, and biological uptake all promote the release of adsorbed pollutants. Organisms that live and feed in sediment are especially vulnerable to contaminants in sediments. Having entered the food chain, contaminants can pass to feeders at higher food (trophic) levels and can accumulate or concentrate in these organisms. Humans can ingest these contaminants by eating fish.

Sediment deposition also can physically alter benthic (bottom) habitats and affect habitat and reproductive potentials for many fish and invertebrates. Sediment sampling should allow all these impact potentials to be assessed.

Collection Techniques

Sediment samples are collected using hand- or winch-operated dredges. Although a wide variety of dredges are available, most operate in the following similar fashion:

1. The device is lowered or pushed through the water column by hand or winch.
2. The device is released to allow closure, either by the attached line or by a weighted messenger that is dropped down the line.
3. The scoops or jaws of the device close either by weight or spring action.
4. The device is retrieved to the surface.

Ideally, the device disturbs the bottom as little as possible and closes fully so that fine particles are not lost. Common benthic sampling devices include the Ponar, Eckman, Peterson, Orange-peel, and Van Veen dredges. When information is needed about how chemical depositions and accumulations have varied through time, sediment cores can be collected with a core sampling device. Very low density or very coarse sediments can be sampled by freeze coring. A thorough description of sediment samplers is included in Klemm et al. (1990).

Sediment sampling techniques are useful for two types of investigations related to stream assessments (1) chemical analysis of sediments and (2) investigation of benthic macroinvertebrate communities. In either type of investigation, sediments from reference stations should be sampled so that they can be compared with sediments in the affected receiving waters. Sediments used for chemical analyses should be removed from the dredge or core samples by scraping back the surface layers of the collected sediment and extracting sediments from the central mass of the collected sample. This helps to avoid possible contamination of the sample by the sample device. Sediment samples for toxicological and chemical examination should be collected following method E 1391 detailed in ASTM

(1991). Sediments for benthic population analyses may be returned in total for cleaning and analysis or may receive a preliminary cleaning in the field using a No. 30 sieve.

Sediment Analyses

There are a variety of sediment analysis techniques, each designed with inherent assumptions about the behavior of sediments and sediment-bound contaminants. An overview of developing techniques is presented in Adams et al. (1992). EPA has evaluated 11 of the methods available for assessing sediment quality (USEPA 1989b). Some of the techniques may help to demonstrate attainment of narrative requirements of some water quality standards. Two of these common analyses are introduced briefly in the following paragraphs.

Bulk sediment analyses analyze the total concentration of contaminants that are either bound to sediments or present in pore water. Results are reported in milligrams or micrograms per kilogram of sediment material. This type of testing often serves as a screening analysis to classify dredged material. Results of bulk testing tend to overestimate the mass of contaminants that will be available for release or for biological uptake because a portion of the contaminants are not biologically available or likely to dissolve.

Elutriate testing estimates the amount of contaminants likely to be released from sediments when mixed with water. In an elutriate test, sediment is mixed with water and then agitated. The standard elutriate test for dredge material mixes four parts water from the receiving water body with one part sediment (USEPA 1990). After vigorous mixing, the sample is allowed to settle before the supernatant is filtered and analyzed for contaminants. This test was designed to estimate the amount of material likely to enter the dissolved phase during dredging; however, it is also useful as a screening test for determining whether further testing should be performed and as a tool for comparing sediments upstream and downstream of potential pollutant sources.

Data Management

All monitoring data should be organized and stored in a readily accessible form. The potentially voluminous and diverse nature of the data, and the variety of individuals who can be involved in collecting, recording, and entering data, can easily lead to the loss of data or the recording of erroneous data. Lost or erroneous data can severely damage the quality of monitoring programs. A sound and efficient data management program for a monitoring program should focus on preventing such problems. This requires that data be managed directly and separately from the activities that use them.

Data management systems include technical and managerial components. The technical components involve selecting appropriate computer equipment and software and designing the database, including data definition, data standardization, and a data dictionary. The managerial components include data entry, data validation and verification, data access, and methods for users to access the data.

To ensure the integrity of the database, it is imperative that data quality be controlled from the point of collection to the time the information is entered into the database. Field and laboratory personnel must carefully enter data into proper spaces on data sheets and avoid transposing numbers. To avoid transcription errors, entries into a database should be made from original data sheets or photocopies. As a preliminary screen for data quality, the database design should include automatic parameter range checking. Values outside the defined ranges should be flagged by the program and immediately corrected or included in a follow-up review of the entered data. For some parameters, it might be appropriate to include automatic checks to disallow duplicate values. Preliminary database files should be printed and verified against the original data to identify errors.

Additional data validation can include expert review of the verified data to identify possible suspicious values. Sometimes, consultation with the individuals responsible for collecting or entering original data is required to resolve problems. After all data are verified and validated, they can be merged into the monitoring program's master database. To prevent loss of data from computer failure, at least one set of duplicate (backup) database files should be maintained at a location other than where the master database is kept.

Quality Assurance and Quality Control (QA/QC)

Quality assurance (QA) is the management process to ensure the quality of data. In the case of monitoring projects, it is managing environmental data collection to ensure the collection of high-quality data. QA focuses on systems, policies, procedures, program structures, and delegation of responsibility that will result in high-quality data. Quality control (QC) is a group of specific procedures designed to meet defined data quality objectives. For example, equipment calibration and split samples are QC procedures. QA/QC procedures are essential to ensure that data collected in environmental monitoring programs are useful and reliable.

The following are specific QA plans required of environmental monitoring projects that receive funding from EPA:

- State and local governments receiving EPA assistance for environmental monitoring projects must complete a quality assurance program plan acceptable to the award official. Guidance for producing the program plan is contained in USEPA (1983d).
- Environmental monitoring projects that receive EPA funding must file a quality assurance project plan, or QAPP, (40 CFR 30.503), the purpose of which is to ensure quality of a specific project. The QAPP describes quality assurance practices designed to produce data of quality sufficient to meet project objectives. Guidance for producing the QAPP (formerly termed the QAPjP) is contained in USEPA (1983e). The plan must ad-

dress the following items:

- Title of project and names of principal investigators.
- Table of contents.
- Project description.
- Project organization and QA/QC responsibility.
- Quality assurance objectives and criteria for determining precision, accuracy, completeness, representativeness, and comparability of data.
- Sampling procedures.
- Sample custody.
- Calibration procedures.
- Analytical procedures.
- Data reduction, validation, and reporting.
- Internal quality control checks.
- Performance and system audits.
- Preventive maintenance procedures.
- Specific routine procedures to assess data precision, accuracy, representativeness, and comparability.
- Corrective action.
- Quality assurance reports.

Sample and Analytical Quality Control

The following quality control techniques are useful in assessing sampling and analytic performance (see also USEPA 1979b, Horwitz et al. 1994):

- *Duplicate samples* are independent samples collected in such a manner that they are equally representative of the contaminants of interest. Duplicate samples, when analyzed by the same laboratory, provide precision information for the entire measurement system, including sample collection, homogeneity, handling, shipping, storage, preparation, and analysis.
- *Split samples* have been divided into two or more portions at some point in the measurement process. Split samples that are divided in the field yield results relating precision to handling, shipping, storage, preparation, and analysis. The split samples may be sent to different laboratories and subjected to the same measurement process to assess interlaboratory variation. Split samples serve an oversight function in assessing the analytical portion of the measurement system, whereas error due to sampling technique may be estimated by analyzing duplicate versions of the same sample.
- *Spiked samples* are those to which a known quantity of a substance is added. The results of spiking a sample in the field are usually expressed as percent recovery of the added material. Spiked samples provide a check of the accuracy of laboratory and analytic procedures.

Sampling accuracy can be estimated by evaluating the results obtained from blanks. The most suitable types of blanks for this appraisal are equipment, field, and trip blanks.

- *Equipment blanks* are samples obtained by running analyte-free water through sample collection equipment, such as a bailer, pump, or auger, after decontamination procedures are completed. These samples are used to determine whether variation is introduced by sampling equipment.
- *Field blanks* are made by transferring deionized water to a sample container at the sampling site. Field blanks test for contamination in the deionized water and contamination introduced through the sampling procedure. They differ from trip blanks, which remain unopened in the field.
- *Trip blanks* test for cross-contamination during transit of volatile constituents, such as many synthetic organic compounds and mercury. For each shipment of sample containers sent to the analytical laboratory, one container is filled with analyte-free water at the laboratory and is sealed. The blanks are transported to the site with the balance of the sample containers and remain unopened. Otherwise, they are handled in the same manner as the other samples. The trip blanks are returned to the laboratory with the samples and are analyzed for the volatile constituents.

Field Quality Assurance

Errors or a lack of standardization in field procedures can significantly decrease the reliability of environmental monitoring data. If required, a quality assurance project plan should be followed for field measurement procedures and equipment. If the QAPP is not formally required, a plan including similar material should be developed to ensure the quality of data collected. Standard operating procedures should be followed when available and should be developed when not.

It is important that quality procedures be followed and regularly examined. For example, field meters can provide erroneous values if they are not regularly calibrated and maintained. Reagent solutions and probe electrolyte solutions have expiration periods and should be refreshed periodically.

7.D Biological Characteristics

Nearly all analytical procedures for assessing the condition of biological resources can be used in stream corridor restoration. Such procedures differ, however, in their scale and focus and in the assumptions, knowledge, and effort required to apply them. These procedures can be grouped into two broad classes—synthetic measures of system condition and analyses based on how well the system satisfies the life history requirements of target species or species groups.

The most important difference between these classes is the logic of how they are applied in managing or restoring a stream corridor system. This chapter focuses on metrics of biological conditions and does not describe, for example, actual field methods for counting organisms.

Synthetic Measures of System Condition

Synthetic measures of system condition summarize some aspect of the structural or functional status of a system at a particular point in time. Complete measurement of the state of a stream corridor system, or even a complete census of all of the species present, is not feasible. Thus, good indicators of system condition are efficient in the sense that they summarize the health of the overall system without having to measure everything about the system.

Use of indicators of system condition in management or restoration depends completely on comparison to values of the indicator observed in other systems or at other times. Thus, the current value of an indicator for a degraded stream corridor can be compared to a previously measured preimpact value for the corridor, a desired future value for the corridor, a value observed at an "unimpacted" reference site, a range of values observed in other systems, or a normative value for that class of stream corridors in a stream classification system. However, the indicator itself and the analysis that establishes the value of the indicator provide no direct information about what has caused the system to have a particular value for the indicator.

Deciding what to change in the system to improve the value of the indicator depends on a temporal analysis in which observed changes in the indicator in one system are correlated with various management actions or on a spatial analysis in which values of the indicator in different systems are correlated with different values of likely controlling variables. In both cases, no more than a general empirical correlation between specific causal factors and the indicator variable is attempted. Thus, management or restoration based on synthetic measures of system condition relies heavily on iterative monitoring of the indicator variable and trial and error, or adaptive management, approaches. For example, an index of species composition based on the presence or absence of a set of sensitive species might be generally correlated with

Stream Visual Assessment Protocol

This is another assessment tool that provides a basic level of stream health evaluation. It is intended to be the first level in a four-part hierarchy of assessment protocols that facilitate planning stream restorations. Scores are assigned by the planners for the following:

- Channel condition
- Hydrologic alteration
- Riparian zone width
- Bank stability
- Canopy cover
- Water appearance
- Nutrient enrichment
- Manure presence
- Salinity
- Barriers to fish movement
- Instream fish cover
- Pools
- Riffle quality
- Invertebrate habitat
- Macroinvertebrates observed

The planning assessment concludes with narratives of the suspected causes of observed problems, as well as recommendations or further steps in the planning process (USDA-NRCS 1998).

water quality, but the index itself provides no information on how water quality should be improved. However, the success of management actions in improving water quality could be tracked and evaluated through iterative measurement of the index.

Synthetic measures of system condition vary along a number of important dimensions that determine their applicability. In certain situations, single species might be good indicators of some aspect of a stream corridor system; in others, community metrics, such as diversity, might be more suitable. Some indicators incorporate physical variables, and others do not. Measurements of processes and rates, such as primary productivity and channel meandering rates, are incorporated into some and not into others. Each of these dimensions must be evaluated relative to the objectives of the restoration effort to determine which, if any, indicator is most appropriate.

Indicator Species

Landres et al. (1988) define an indicator species as an organism whose characteristics (e.g., presence or absence, population density, dispersion, reproductive success) are used as an index of attributes too difficult, inconvenient, or expensive to measure for other species or environmental conditions of interest. Ecologists and management agencies have used aquatic and terrestrial indicator species for many years as assessment tools, the late 1970s and early 1980s being a peak interest period. During that time, Habitat Evaluation Procedures (HEP) were developed by the U.S. Fish and Wildlife Service, and the U.S. Forest Service's use of management indicator species was mandated by law with passage of the National Forest Management Act in 1976. Since that time, numerous authors have expressed concern about the ability of indicator species to meet the expectations expressed in the above definition. Most notably, Landres et al. (1988) critically evaluated the use of vertebrate species as ecological indicators and suggested that rigorous justification and evaluation are needed before the concept is used. The discussion of indicator species below is largely based on their paper.

The Good and Bad of Indicator Species

Indicator species have been used to predict environmental contamination, population trends, and habitat quality; however, their use in evaluating water quality is not covered in this section. The assumptions implicit in using indicators are that if the habitat is suitable for the indicator it is also suitable for other species (usually in a similar ecological guild) and that wildlife populations reflect habitat conditions. However, because each species has unique life requisites, the relationship between the indicator and its guild may not be completely reliable, although the literature is inconsistent in this regard (see Riparian Response Guilds subsection below). It is also difficult to include all the factors that might limit a population when selecting a group of species that an indicator is expected to represent. For example, similarities in breeding habitat between the indicator and its associates might appear to group species when in fact differences in predation rates, disease, or winter habitat actually limit populations.

Some management agencies use vertebrate indicators to track changes in habitat condition or to assess the influence of habitat alteration on selected species. Habitat suitability indices and other habitat models are often used for this purpose, though the metric chosen to measure a species' response to its habitat can influence the outcome of the investigation. As Van Horne (1983) pointed out, density and other abundance metrics may be misleading indicators of habitat quality. Use of diversity and other indices to estimate habitat quality also creates problems when the variation in measures yields an average value for an index that might not represent either extreme.

Selecting Indicators

Landres et al. (1988) suggest that if the decision is made to use indicators, then several factors are important to consider in the selection process:

- Sensitivity of the species to the environmental attribute being evaluated. When possible, data that suggest a cause-and-effect relationship are preferred to correlates (to ensure the indicator reflects the variable of interest and not a correlate).
- Indicator accurately and precisely responds to the measured effect. High variation statistically limits the ability to detect effects. Generalist species do not reflect change as well as more sensitive endemics. However, because specialists usually have lower populations, they might not be the best for cost-effective sampling. When the goal of monitoring is to evaluate on-site conditions, using indicators that occur only within the site makes sense. However, although permanent residents may better reflect local conditions, the goal of many riparian restoration efforts is to provide habitat for neotropical migratory birds. In this case, residents such as cardinals or woodpeckers might not serve as good indicators for migrating warblers.

- Size of the species home range. If possible, the home range should be larger than that of other species in the evaluation area. Management agencies often are forced to use high-profile game or threatened and endangered species as indicators. Game species are often poor indicators simply because their populations are highly influenced by hunting mortality, which can mask environmental effects. Species with low populations or restrictions on sampling methods, such as threatened and endangered species, are also poor indicators because they are difficult to sample adequately, often due to budget constraints. For example, Verner (1986) found that costs to detect a 10 percent change in a randomly sampled population of pileated woodpeckers would exceed a million dollars per year.
- Response of an indicator species to an environmental stressor cannot be expected to be consistent across varying geographic locations or habitats without corroborative research.

Riparian Response Guilds

Vertebrate response guilds as indicators of restoration success in riparian ecosystems may be a valuable monitoring tool but should be used with the same cautions presented above. Croonquist and Brooks (1991) evaluated the effects of anthropogenic disturbances on small mammals and birds along Pennsylvania waterways. They evaluated species in five different response guilds, including wetland dependency, trophic level, species status (endangered, recreational, native, exotic), habitat specificity, and seasonality (birds).

They found that community coefficient indices were better indicators than species richness. The habitat specificity and seasonality response guilds for birds were best able to distinguish those species sensitive to disturbance from those which were not affected or were benefited. Neotropical migrants and species with specific habitat requirements were the best predictors of disturbance. Edge and exotic species were greater in abundance in the disturbed habitats and might serve as good indicators there. Seasonality analysis showed migrant breeders were more common in undisturbed areas, which, as suggested by Verner (1984), indicates the ability of guild analysis to distinguish local impacts. Mammalian response guilds did not exhibit any significant sensitivity to disturbance and were considered unsuitable as indicators.

In contrast, Mannan et al. (1984) found that in only one of the five avian guilds tested was the density of birds consistent across managed and undisturbed forests. In other words, population response to restoration might not be consistent across different indicator guilds. Also, periodically monitoring restoration initiatives is necessary to document when, during the recovery stage, the more sensitive species outcompete generalists.

Aquatic Invertebrates

Aquatic invertebrates have been used as indicators of stream and riparian health for many years. Perhaps more than other taxa, they are closely tied to both aquatic and riparian habitat. Their life cycles usually include periods in and out of the water, with ties to riparian vegetation for feeding, pupation, emergence, mating, and egg laying (Erman 1991).

It is often important to look at the entire assemblage of aquatic invertebrates as an indicator group. Impacts to a stream often decrease diversity but might increase the abundance of some species, with the size of the first species to be affected often larger (Wallace and Gurtz 1986). In summary, a good indicator species should be low on the food chain to respond quickly, should have a narrow tolerance to change, and should be a native species (Erman 1991).

Diversity and Related Indices

Biological diversity refers to the number of species in an area or region and includes a measure of the variety of species in a community that takes into account the relative abundance of each species (Ricklefs 1990). When measuring diversity, it is important to clearly define the biological objectives, stating exactly what attributes of the system are of concern and why (Schroeder and Keller 1990). Different measures of diversity can be applied at various levels of complexity, to different taxonomic groups, and at distinct spatial scales. Several factors should be considered in using diversity as a measure of system condition for stream corridor restoration.

Levels of Complexity

Diversity can be measured at several levels of complexity—genetic, population/species, community/ecosystem, and landscape (Noss 1994). There is no single correct level of complexity to use because different scientific or management issues are focused on different levels (Meffe et al. 1994). The level of complexity chosen for a specific stream corridor restoration initiative should be determined based on careful consideration of the biological objectives of the project.

Subsets of Concern

Overall diversity within any given level of complexity may be of less concern than diversity of a particular subset of species or habitats. Measures of overall diversity include all of the elements of concern and do not provide information about the occurrence of specific elements. For example, measures of overall species diversity do not provide information about the presence of individual species or species groups of management concern.

Any important subsets of diversity should be described in the process of setting biological objectives. At the community level, subsets of species of interest might include native, endemic, locally rare or threatened, specific guilds (e.g., cavity users), or taxonomic groups (e.g., amphibians, breeding birds, macroinvertebrates). At the terrestrial landscape level, subsets of diversity could include forest types or seral stages (Noss 1994). Thus, for a specific stream corridor project, measurement of diversity may be limited to a target group of special concern. In this manner, comparison of diversity levels becomes more meaningful.

Spatial Scale

Diversity can be measured within the bounds of a single community, across community boundaries, or in large areas encompassing many communities. Diversity within a relatively homogeneous community is known as alpha diversity. Diversity between communities, described as the amount of differentiation along habitat gradients, is termed beta diversity. The total diversity across very large landscapes is gamma diversity. Noss and Harris (1986) note that management for alpha diversity may increase local species richness, while the regional landscape (gamma diversity) may become more homogeneous and less diverse overall. They recommend a goal of maintaining the regional species pool in an approximately natural relative abundance pattern. The specific size of the area of concern should be defined when diversity objectives are established.

Measures of Diversity

Magurran (1988) describes three main categories of diversity measures—richness indices, abundance models, and indices based on proportional abundance. Richness indices are measures of the number of species (or other element of diversity) in a specific sampling unit and are the most widely used indices (Magurran 1988). Abundance models account for the evenness (equitability) of distribution of species and fit various distributions to known models, such as the geometric series, log series, lognormal, or broken stick. Indices based on the proportional abundance of species combine both richness and evenness into a single index. A variety of such indices exist, the most common of which is the Shannon-Weaver diversity index (Krebs 1978):

$$H = -\Sigma p_i \log_e p_i$$

where

H = index of species diversity

S = number of species

p_i = proportion of total sample belonging to the i^{th} species

Results of most studies using diversity indices are relatively insensitive to the particular index used (Ricklefs 1979). For example, bird species diversity indices from 267 breeding bird censuses were highly correlated ($r = 0.97$) with simple counts of bird species richness (Tramer 1969). At the species level, a simple measure of richness is most often used in conservation biology studies because the many rare species that characterize most systems are generally of greater interest than the common species that dominate in diversity indices and because accurate population density estimates are often not available (Meffe et al. 1994).

Simple measures of species richness, however, are not sensitive to the actual species composition of an area. Similar richness values in two different areas may represent very different sets of species. The usefulness of these measures can be increased by considering specific subsets of species of most concern, as mentioned above. Magurran (1988) recommends going beyond the use of a single diversity measure and examining the shape of the species abundance distribution as well. Breeding bird census data from an 18-hectare (ha) riparian deciduous forest habitat in Ohio (Tramer 1996) can be used to illustrate these different methods of presentation **(Figure 7.36)**. Breeding bird species richness in this riparian habitat was 38.

Pielou (1993) recommends the use of three indices to adequately assess diversity in terrestrial systems:

- A measure of plant species diversity.
- A measure of habitat diversity.
- A measure of local rarity.

Other indices used to measure various aspects of diversity include vegetation measures, such as foliage height diversity (MacArthur and MacArthur 1961), and landscape measures, such as fractal dimension, fragmentation indices, and juxtaposition (Noss 1994).

Related Integrity Indices

Karr (1981) developed the Index of Biotic Integrity to assess the diversity and health of aquatic communities. This index is designed to assess the present status of the aquatic community using fish community parameters related to species composition, species richness, and ecological factors. Species composition and richness parameters may include the presence of intolerant species, the richness and composition of specific species groups (e.g., darters), or the proportion of specific groups (e.g., hybrid individuals). Ecological parameters may include the proportion of top carnivores, number of individuals, or proportion with disease or other anomalies. Key parameters are devel-

Figure 7.36: Breeding bird census data.
Species abundance curve in a riparian deciduous forest habitat.
From Tramer 1996.

Species Sequence	Species	Abundance in 18-ha Plot
1	American robin	18.5
2	House wren	13
3	Gray catbird	10.5
4	Song sparrow	9.5
5	Northern cardinal	7.5
6	Baltimore oriole	7
7	Warbling vireo	6
8	Wood thrush	4.5
9	Common grackle	4.5
10	Eastern wood-pewee	4
11	Red-eyed vireo	4
12	Indigo bunting	4
13	Red-winged blackbird	4
14	Mourning dove	3
15	Northern flicker	3
16	Blue jay	3
17	Tufted titmouse	3
18	White-breasted nuthatch	3
19	American redstart	3
20	Rose-breasted grosbeak	3
21	Downy woodpecker	2
22	Great crested flycatcher	2
23	Black-capped chickadee	2
24	Carolina wren	2
25	European starling	2
26	Yellow warbler	2
27	Brown-headed cowbird	2
28	American goldfinch	2
29	Wood duck	1
30	Ruby-throated hummingbird	1
31	Red-bellied woodpecker	1
32	Hairy woodpecker	1
33	Tree swallow	1
34	Blue-gray gnatcatcher	1
35	Prothonotary warbler	1
36	Common yellowthroat	1
37	Eastern phoebe	1
38	N. rough-winged swallow	1

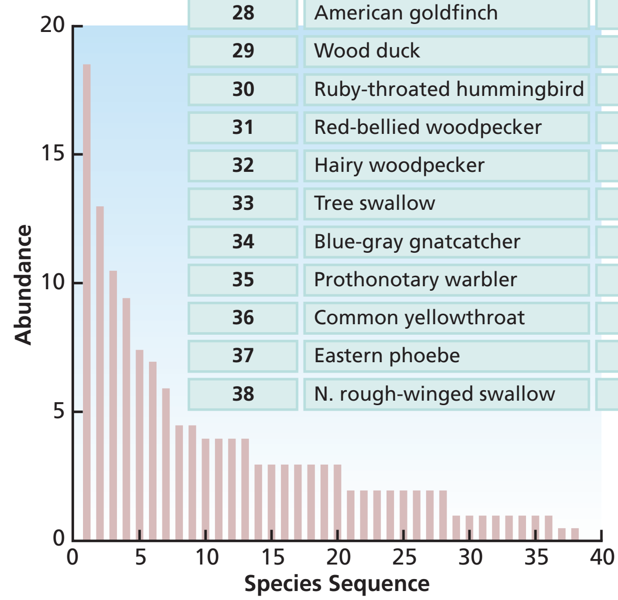


Table 7.8: Five tiers of the rapid bioassessment protocols.
RBPs are used to conduct cost-effective biological assessments.
From Plafkin et al. 1989.

Level or Tier	Organism Group	Relative Level of Effort	Level of Taxonomy/ Where Performed	Level of Expertise Required
I	Benthic invertebrates	Low; 1-2 hr per site (no standardized sampling)	Order, family/field	One highly-trained biologist
II	Benthic invertebrates	Intermediate; 1.5-2.5 hr per site (all taxonomy performed in field)	Family/field	One highly-trained biologist and one technician
III	Benthic invertebrates	Most rigorous; 3-5 hr per site (2-3 hr of total are for lab taxonomy)	Genus or species/laboratory	One highly-trained biologist and one technician
IV	Fish	Low; 1-3 hr per site (no fieldwork involved)	Not applicable	One highly-trained biologist
V	Fish	Most rigorous; 2-7 hr per site (1-2 hr per site are for data analysis)	Species/field	One highly-trained biologist and 1-2 technicians

oped for the stream system of interest, and each parameter is assigned a rating. The overall rating of a stream is used to evaluate the quality of the aquatic biota.

Rapid Bioassessment

Rapid bioassessment techniques are most appropriate when restoration goals are nonspecific and broad, such as improving the overall aquatic community or establishing a more balanced and diverse community in the stream corridor. Bioassessment often refers to use of biotic indices or composite analyses, such as those used by Ohio EPA (1990), and rapid bioassessment protocols (RBP), such as those documented by Plafkin et al. (1989). Ohio EPA evaluates biotic integrity by using an invertebrate community index (ICI) that emphasizes structural attributes of invertebrate communities and compares the sample community with a reference or control community. The ICI is based on 10 metrics that describe different taxonomic and pollution tolerance relationships within the macroinvertebrate community. The RBP established by USEPA (Plafkin et al. 1989) were developed to provide states with the technical information necessary for conducting cost-effective biological assessments. The RBP are divided into five sets of protocols (RBP I to V), three for macroinvertebrates and two for fish (**Table 7.8**).

Algae

Although not detailed by Plafkin et al. (1989), algal communities are useful for bioassessment. Algae generally have short life spans and rapid reproduction rates, making them useful for evaluating short-term impacts. Sampling impacts are minimal to resident biota, and collection requires little effort. Primary productivity of algae is affected by physical and chemical impairments. Algal communities are sensitive to some pollutants that might not visibly affect other aquatic communities. Algal communities can be examined for indicator species, diversity indices, taxa richness, community respiration, and colonization rates. A variety of nontaxonomic evaluations, such as biomass and chlorophyll, may be used and are summarized in Weitzel (1979). Rodgers et al. (1979) describe functional measurements of algal communities, such as primary productivity and community respiration, to evaluate the effects of nutrient enrichment.

Although collecting algae in streams requires little effort, identifying for metrics, such as diversity indices and taxa richness, may require considerable effort. A great deal of effort may be expended to document diurnal and seasonal variations in productivity.

Benthic Macroinvertebrates

The intent of the benthic rapid bioassessment is to evaluate overall biological condition, optimizing the use of the benthic community's capacity to reflect integrated environmental effects over time. Using benthic macroinvertebrates is advantageous for the following reasons:

- They are good indicators of localized conditions.
- They integrate the effects of short-term environmental variables.
- Degraded conditions are easily detected.
- Sampling is relatively easy.
- They provide food for many fish of commercial or recreational importance.
- Macroinvertebrates are generally abundant.
- Many states already have background data.

As indicated above, the RBP are divided into three sets of protocols (RBP I to III) for macroinvertebrates. RBP I is a "screening" or reconnaissance-level analysis used to discriminate obviously impaired and nonimpaired sites from potentially affected areas requiring further investigation. RBP II and III use a set of metrics based on taxon tolerance and community structure similar to the ICI used by the state of Ohio. Both are more labor-intensive than RBP I and incorporate field sampling. RBP II uses family-level taxonomy to determine the following set of metrics used in describing the biotic integrity of a stream:

- Taxa richness.
- Hilsenhoff biotic index (Hilsenhoff 1988).
- Ratio of scrapers to filtering collectors.
- Ratio of Ephemeroptera/Plecoptera/Trichoptera (EPT) and chironomid abundances.
- Percent contribution of dominant taxa.
- EPT index.
- Community similarity index.
- Ratio of shredders to total number of individuals.

RBP III further defines the level of biotic impairment and is essentially an intensified version of RBP II that uses species-level taxonomy. As with ICI, the RBP metrics for a site are compared to metrics from a control or reference site.

Fish

Hocutt (1981) states "perhaps the most compelling ecological factor is that structurally and functionally diverse fish communities both directly and indirectly provide evidence of water quality in that they incorporate all the local environmental perturbations into the stability of the communities themselves."

Figure 7.37: Fish samples.
Water quality standards are often characterized in terms of fisheries.

The advantages of using fish as bioindicators are as follows:

- They are good indicators of long-term effects and broad habitat conditions.
- Fish communities represent a variety of trophic levels.
- Fish are at the top of the aquatic food chain and are consumed by humans.
- Fish are relatively easy to collect and identify.
- Water quality standards are often characterized in terms of fisheries.
- Nearly one-third the endangered vertebrate species and subspecies in the United States are fish.

The disadvantages of using fish as bioindicators are as follows:

- The cost.
- Statistical validity may be hard to attain.
- It is difficult to interpret findings.

Electrofishing is the most commonly used field technique. Each collecting station should be representative of the study reach and similar to other reaches sampled; effort between reaches should be equal. All fish species, not just game species, should be collected for the fish community assessment (**Figure 7.37**). Karr et al. (1986) used 12 biological metrics to assess biotic integrity using taxonomic and trophic composition and condition and abundance of fish. Although the Index of Biological Integrity (IBI) developed by Karr was designed for small midwestern streams, it has been modified for many regions of the country and for use in large rivers (see Plafkin et al. 1989).

Establishing a Standard of Comparison

With stream restoration activities, it is important to select a desired end condition for the proposed management action. A predetermined standard of comparison provides a benchmark against which to measure progress. For example, if the chosen diversity measure is native species richness, the standard of comparison might be the maximum expected native species richness for a defined geographic area and time period.

Historical conditions in the region should be considered when establishing a standard of comparison. If current conditions in a stream corridor are degraded, it may be best to establish the standard at a period in the past that represented more natural or desired conditions. Knopf (1986) notes that for certain western streams, historical diversity might have been less than current due to changes in hydrology and encroachment of native and exotic riparian vegetation in the floodplain. Thus, it is important to agree on what conditions are desired prior to establishing the standard of comparison. In addition, the geographic location and size of the area should be considered. Patterns of diversity vary with geographic location, and larger areas are typically more diverse than smaller areas.

The IBI is scaled to a standard of comparison determined through either professional judgment or empirical data, and such indices have been developed for a variety of streams (Leonard and Orth 1986, Bramblett and Fausch 1991, Lyons et al. 1996).

Evaluating the Chosen Index

For a hypothetical stream restoration initiative, the following biological diversity objective might be developed. Assume that a primary concern in the area is conserving native amphibian species and that 30 native species of amphibians have been known to occur historically in the 386 mi^2 watershed. The objective could be to manage the stream corridor to provide and maintain suitable habitat for the 30 native amphibian species.

Stream corridor restoration efforts must be directed toward those factors that can be managed to increase diversity to the desired level. Those factors might be the physical and structural features of the stream corridor or possibly the presence of an invasive species in the community. Knowledge of the important factors can be obtained from existing literature and from discussions with local and regional experts.

Diversity can be measured directly or predicted from other information. Direct measurement requires an actual inventory of the element of diversity, such as counting the amphibian species in the study area. The IBI requires sampling fish populations to determine the number and composition of fish species. Measures of the richness of a particular animal group require counts. Determining the number of species in a community is best accomplished with a long-term effort because there can be much variation over short periods. Variation can arise from observer differences, sampling design, or temporal variation in the presence of species.

Direct measures of diversity are most helpful when baseline information is available for comparing different sites. It is not possible, however, to directly measure certain attributes, such as species richness or the population level of various species, for various future conditions. For example, the IBI cannot be directly computed for a predicted stream corridor condition, following management action.

Predictions of diversity for various future conditions, such as with restoration or management, require the use of a predictive model. Assume the diversity objective for a stream corridor restoration effort is to maximize native amphibian species richness. Based on knowledge of the life history of the species, including requirements for habitat, water quality, or landscape configuration, a plan can be developed to restore a stream corridor to meet these needs. The plan could include a set of criteria or a model to describe the specific features that should be

Table 7.9: Selected riverine and riparian classification systems. *Classification systems are useful in characterizing biological conditions.*

Classification System	Subject	Geographic Domain	Citation
Riparian vegetation of Yampa, San Miguel/Dolores River Basins	Plant communities	Colorado	Kittel and Lederer (1993)
Riparian and scrubland communities of Arizona and New Mexico	Plant communities	Arizona and New Mexico	Szaro (1989)
Classification of Montana riparian and wetland sites	Plant communities	Montana	Hansen et al. (1995)
Integrated riparian evaluation guide	Hydrology, geomorphology, soils, vegetation	Intermountain	U.S. Forest Service (1992)
Streamflow cluster analysis	Hydrology with correlations to fish and invertebrates	National	Pott and Ward (1989)
River Continuum	Hydrology, stream order, water chemistry, aquatic communities	International, national	Vannote et al. (1980)
World-wide stream classification	Hydrology, water chemistry, substrate, vegetation	International	Pennak (1971)
Rosgen's river classification	Hydrology, geomorphology: stream and valley types	National	Rosgen (1996)
Hydrogeomorphic wetland classification	Hydrology, geomorphology, vegetation	National	Brinson (1993)
Recovery classes following channelization	Hydrology, geomorphology, vegetation	Tennessee	Hupp (1992)

included to maximize amphibian richness. Examples of indirect methods to assess diversity include habitat models (Schroeder and Allen 1992, Adamus 1993) and cumulative impact assessment methods (Gosselink et al. 1990, Brooks et al. 1991).

Predicting diversity with a model is generally more rapid than directly measuring diversity. In addition, predictive methods provide a means to analyze alternative future conditions before implementing specific restoration plans. The reliability and accuracy of diversity models should be established before their use.

Classification Systems

Classification is an important component of many of the scientific disciplines relevant to stream corridors—hydrology, geomorphology, limnology, plant and animal ecology. **Table 7.9** lists some of the classification systems that might be useful in identifying and planning riverine restoration activities. It is not the intent of this section to exhaustively review all classification schemes or to present a single recommended classification system. Rather, we focus on some of the principal distinctions among classification systems and factors to consider in the use of classification systems for restoration planning, particularly in the use of a classification system as a measure of biological condition. It is likely that multiple systems will be useful in most actual riverine restoration programs.

The common goal of classification systems is to organize variation. Important dimensions in which riverine classification systems differ include the following:

- *Geographic domain*. The range of sites being classified varies from rivers of the world to local differences in the composition and characteristics of patches within one reach of a single river.

- *Variables considered.* Some classifications are restricted to abiotic variables of hydrology, geomorphology, and aquatic chemistry. Other community classifications are restricted to biotic variables of species composition and abundance of a limited number of taxa. Many classifications include both abiotic and biotic variables. Even purely abiotic classification systems are relevant to biological evaluations because of the important correlations (e.g., the whole concept of physical habitat) between abiotic structure and community composition.
- *Incorporation of temporal relations.* Some classifications focus on describing correlations and similarities across sites at one, perhaps idealized, point in time. Other classifications identify explicit temporal transitions among classes, for example, succession of biotic communities or evolution of geomorphic landforms.
- *Focus on structural variation or functional behavior.* Some classifications emphasize a parsimonious description of observed variation in the classification variables. Others use classification variables to identify types with different behaviors. For example, a vegetation classification can be based primarily on patterns of species co-occurrence, or it can be based on similarities in functional effect of vegetation on habitat value.
- *The extent to which management alternatives or human actions are explicitly considered as classification variables.* To the extent that these variables are part of the classification itself, the classification system can directly predict the result of a management action. For example, a vegetation classification based on grazing intensity would predict a change from one class of vegetation to another class based on a change in grazing management.

Use of Classification Systems in Restoring Biological Conditions

Restoration efforts may apply several national and regional classification systems to the riverine site or sites of interest because these are efficient ways to summarize basic site description and inventory information and they can facilitate the transference of existing information from other similar systems.

Most classification systems are generally weak at identifying causal mechanisms. To varying degrees, classification systems identify variables that efficiently describe existing conditions. Rarely do they provide unequivocal assurance about how variables actually cause the observed conditions. Planning efficient and effective restoration actions generally requires a much more mechanistic analysis of how changes in controllable variables will cause changes toward desired values of response variables. A second limitation is that application of a classification system does not substitute for goal setting or design. Comparison of the degraded system to an actual unimpacted reference site, to the ideal type in a classification system, or to a range of similar systems can provide a framework for articulating the desired state of the degraded system. However, the desired state of the system is a management objective that ultimately comes from outside the classification of system variability.

Analyses of Species Requirements

Analyses of species requirements involve explicit statements of how variables interact to determine habitat or how well a system provides for the life requisites of fish and wildlife species. Complete specification of relations between all relevant variables and all species in a stream corridor system is not possible. Thus, analyses based on species requirements focus on one or more target species or groups of species. In a simple case, this type of analysis may be based on an explicit statement of the physical factors that distinguish good habitat for a species (places where it is most likely to be found or where it best reproduces) from poor habitat (places where it is unlikely to be found or reproduces poorly). In more complicated cases, such approaches incorporate variables beyond those of purely physical habitat, including other species that provide food or biotic structure, other species as competitors or predators, or spatial or temporal patterns of resource availability.

Analyses based on species requirements differ from synthetic measures of system condition in that they explicitly incorporate relations between “causal” variables and desired biological attributes. Such analyses can be used directly to decide what restoration actions will achieve a desired result and to evaluate the likely consequences of a proposed restoration action. For example, an analysis using the habitat evaluation procedures might identify mast production (the accumulation of nuts from a productive fruiting season which serves as a food source for animals) as a factor limiting squirrel populations. If squirrels are a species of concern, at least some parts of the stream restoration effort should be directed toward increasing mast production. In practice, this logical power is often compromised by incomplete knowledge of the species habitat requirements.

The complexity of these methods varies along a number of important dimensions, including prediction of habitat suitability versus population numbers, analysis for a single place and single time versus a temporal sequence of spatially complex requirements, and analysis for a single target species versus a set of target species involving trade-offs. Each of these dimensions must be carefully considered in selecting an analysis procedure appropriate to the problem at hand.

The Habitat Evaluation Procedures (HEP)

Habitat evaluation procedures (HEP) can be used for several different types of habitat studies, including impact assessment, mitigation, and habitat management. HEP provides information for two general types of habitat comparisons—the relative value of different areas at the same point in time and the relative value of the same area at different points in time. Potential changes in wildlife (both aquatic and terrestrial) habitat due to proposed projects are characterized by combining these two types of comparisons.

Basic Concepts

HEP is based on two fundamental ecological principles—habitat has a definable carrying capacity, or suitability, to support or produce wildlife populations (Fretwell and Lucas 1970), and the suitability of habitat for a given wildlife species can be estimated using measurements of vegetative, physical, and chemical traits of the habitat. The suitability of a habitat for a given species is described by a habitat suitability index (HSI) constrained between 0 (unsuitable habitat) and 1 (optimum habitat). HSI models have been developed and published by the U.S. Fish and Wildlife Service (Schamberger et al. 1982; Terrell and Carpenter, in press), and USFWS (1981) provides guidelines for use in developing HSI models for specific projects. HSI models can be developed for many of the previously described metrics, including species, guilds, and communities (Schroeder and Haire 1993).

The fundamental unit of measure in HEP is the Habitat Unit, computed as follows:

$$HU = AREA \times HSI$$

where HU is the number of habitat units (units of area), AREA is the areal extent of the habitat being described (units of area), and HSI is the index of suitability of the habitat (unitless). Conceptually, an HU integrates the quantity and quality of habitat into a single measure, and one HU is equivalent to one unit of optimal habitat.

Use of HEP to Assess Habitat Changes

HEP provides an assessment of the net change in the number of HUs attributable to a proposed future action, such as a stream restoration initiative. A HEP application is essentially a two-step process—calculating future HUs for a particular project alternative and calculating the net change as compared to a base condition.

The steps involved in using and applying HEP to a management project are outlined in detail in USFWS (1980a). However, some early planning decisions often are given little attention although they may be the most important part of a HEP study. These initial decisions include forming a study team, defining the study boundaries, setting study objectives, and selecting the evaluation species. The study team usually consists of individuals representing different agencies and viewpoints. One member of the team is generally from the lead project planning agency and other members are from resources agencies with an interest in the resources that would be affected.

One of the first tasks for the team is to delineate the study area boundaries. The study area boundaries should be drawn to include any areas of direct impact, such as a flood basin for a new reservoir, and any areas of secondary impact, such as a downstream river reach that might have an altered flow, increased turbidity, or warmer temperature, or riparian or upland areas subject to land use changes as a result of an increased demand on recreational lands. Areas such as an upstream spawning ground that are not contiguous to the primary impact site also might be affected and therefore should be included in the study area.

The team also must establish project objectives, an often neglected aspect of project planning. Objectives should state what is to be accomplished in the project and specify an endpoint to the project. An integral aspect of objective setting is selecting evaluation species, the specific wildlife resources of concern for which HUs will be computed in the HEP analysis. These are often individual species, but they do not have to be. Depending on project objectives, species' life stages (e.g., juvenile salmon), species' life requisites (e.g., spawning habitat), guilds (e.g., cavity-nesting birds), or communities (e.g., avian richness in riparian forests) can be used.

REVERSE | FAST FORWARD

Review Chap. 5's Supporting Analysis for Selecting Restoration Alternatives

Instream Flow Incremental Methodology

The Instream Flow Incremental Methodology (IFIM) is an adaptive system composed of a library of models that are linked to describe the spatial and temporal habitat features of a given river. IFIM is described in Chapter 5 under *Supporting Analysis for Selecting Restoration Alternatives*.

Physical Habitat Simulation

The Physical Habitat Simulation (PHABSIM) model was designed by the U.S. Fish and Wildlife Service primarily for instream flow analysis (Bovee 1982). It represents the habitat evaluation component of a larger instream flow incremental methodology for incorporating fish habitat consideration into flow management, presented in Chapter 5. PHABSIM is a collection of computer programs that allows evaluation of available habitat within a study reach for various life stages of different fish species. The two basic components of the model are hydraulic simulation (based on field-measured cross-sectional data) and several standard hydraulic methods for predicting water surface elevations and velocities at unmeasured discharges (e.g., stage vs. discharge relations, Manning's equation, step-backwater computations). Habitat simulation integrates species and life-stage-specific habitat suitability curves for water depth, velocity, and substrate with the hydraulic data. Output is a plot of weighted usable area (WUA) against discharge for the species and life stages of interest. (**Figure 7.38**)

The stream hydraulic component predicts depths and water velocities at unobserved flows at specific locations on a cross section of a stream. Field measurements of depth, velocity, substrate material, and cover at specific sampling points on a cross section are taken at different observable flows. Hydraulic measurements, such as water surface elevations, also are collected during the field inventory. These data are used to calibrate the hydraulic simulation models. The

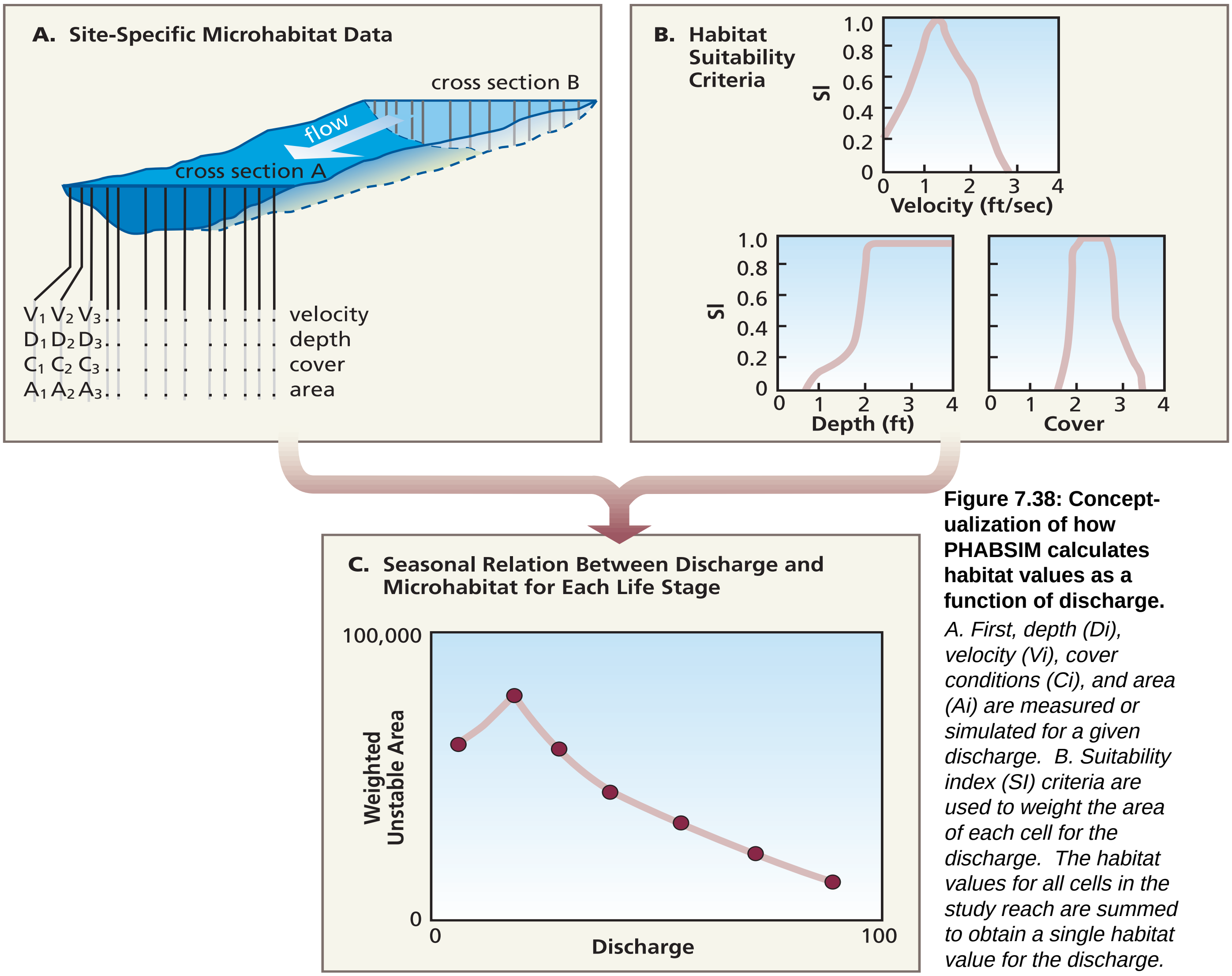


Figure 7.38: Conceptualization of how PHABSIM calculates habitat values as a function of discharge.

A. First, depth (Di), velocity (Vi), cover conditions (Ci), and area (Ai) are measured or simulated for a given discharge. B. Suitability index (SI) criteria are used to weight the area of each cell for the discharge. The habitat values for all cells in the study reach are summed to obtain a single habitat value for the discharge. C. The procedure is repeated for a range of discharges.

Modified from Nestler et al. 1989.

models then are used to predict depths and velocities at flows different from those measured.

The habitat component weights each stream cell using indices that assign a relative value between 0 and 1 for each habitat attribute (depth, velocity, substrate material, cover), indicating how suitable that attribute is for the life stage under consideration. These attribute indices are usually termed habitat suitability indices and are developed from direct observations of the attributes used most often by a life stage, from expert opinion about what the life requisites are, or a combination. Various approaches are taken to factor assorted biases out of these suitability data, but they remain indices that are used as weights of suitability. In the last step of the habitat component, hydraulic estimates of depth and velocity at different flow levels are combined with the suitability values for those attributes to weight the area of each cell at the simulated flows. The weighted values for all cells are summed to produce the WUA.

There are many variations on the basic approach outlined above, with specific analyses tailored for different water management phenomena (such as hydropeaking and unique spawning habitat needs), or for special habitat needs (such as bottom velocity instead of mean column velocity) (Milhous et al. 1989). However, the fundamentals of hydraulic and habitat modeling remain the same, resulting in a WUA versus discharge function. This function should be combined with the appropriate hydrologic time series (water availability) to develop an idea of what life states might be affected by a loss or gain of available habitat and at what time of the year. Time series analysis plays this role and also factors in any physical and institutional constraints on water management so that alternatives can be evaluated (Milhous et al. 1990).

Several things must be remembered about PHABSIM. First, it provides an index to microhabitat availability; it is not a measure of the habitat actually used by aquatic organisms. It can be used only if the species under consideration exhibit documented preferences for depth, velocity, substrate material, cover, or other predictable microhabitat attributes in a specific environment of competition and predation. The typical application of PHABSIM assumes relatively steady flow conditions such that depths and velocities are comparably stable within the chosen time step. PHABSIM does not predict the effects of flow on channel change. Finally, the field data and computer analysis requirements can be relatively large.

Two-dimensional Flow Modeling

Concern about the simplicity of the one-dimensional hydraulic models used in PHABSIM has led to current research interest in the use of more sophisticated two-dimensional hydraulic models to simulate physical conditions of depth and velocity for use in fish habitat analysis. A two-dimensional hydraulic model can be spatially adjusted to represent the scale of aquatic habitat and the variability of other field data. For example, the physical relationship between different aquatic habitat types is often a key parameter when considering fish habitat use. The spatial nature of two-dimensional flow modeling allows for the analysis of these relationships. The model can also consider the drying and wetting of intermittent stream channels.

Leclerc et al. (1995) used two-dimensional flow modeling to study the effect of a water diversion on the habitat of juvenile Atlantic salmon (*Salmo salar*) in the Moisie River in Quebec, Canada. Average model error was reduced when compared with traditional one-dimensional models. Output from the two-dimensional modeling was combined with habitat suitability indexes with finite element calculation techniques. Output from the analysis included maps displaying the spatial distribution of depth, velocity, and habitat suitability intervals.

Physical data collection for this modeling tool is intensive. Channel contour and bed material mapping is required along with discharge relationships and the upstream and downstream boundaries of each study reach. Velocity and water-surface measurements for various discharges are required for model calibration. Two-dimensional modeling does not address all of the issues related to hydrodynamics and flow modeling. Mobile bed systems and variability in Manning's coefficient are still problematic using this tool (Leclerc et al. 1995). Moderate to large rivers with a stable bedform are most suited to this methodology.

Riverine Community Habitat Assessment and Restoration Concept Model (RCHARC)

Another modeling approach to aquatic habitat restoration is the Riverine Community Habitat Assessment and Restoration (RCHARC) concept. This model is based on the assumption that aquatic habitat in a restored stream reach will best mimic natural conditions if the bivariate frequency distribution of depth and velocity in the subject channel is similar to a reference reach with good aquatic habitat. Study site and reference site data can be measured or calculated using a computer model. The similarity of the proposed design and reference reach is expressed with three-dimensional graphs and statistics (Nestler et al. 1993, Abt 1995). RCHARC has been used as the primary tool for environmental analysis on studies of flow management for the Missouri River and the Alabama-Coosa-Tallapoosa Apalachicola-Chattahoochee-Flint Basin.

Time Series Simulations

A relatively small number of applications have been made of time series simulations of fish population or individual fish responses to riverine habitat changes. Most of these have used PHABSIM to accomplish hydraulic model development and validation and hydraulic simulation, but some have substituted time-series simulations of individual or population responses for habitat suitability curve development and validation, and habitat suitability modeling. PHABSIM quantifies the relationship of hydraulic estimates (depth and velocity) and measurements (substrate and cover) with habitat suitability for target fish and invertebrate life stages or water-related recreation suitability. It is useful when relatively steady flow is the major determinant controlling riverine resources. Use of PHABSIM is generally limited to river systems in which dissolved oxygen, suspended sediment, nutrient loading, other chemical aspects of water quality, and interspecific competition do not place the major limits on populations of interest. These limitations to the use of PHABSIM can be abated or removed with models that simulate response of individual fish or fish populations.

Individual-based Models

The Electric Power Research Institute (EPRI) program on compensatory mechanisms in fish populations (CompMech) has the objective of improving predictions of fish population response to increased mortality, loss of habitat, and release of toxicants (EPRI 1996). This technique has been applied by utilities and resource management agencies in assessments involving direct mortality due to entrainment, impingement, or fishing; instream flow; habitat alteration (e.g., thermal discharge, water-level fluctuations, water diversions, exotic species); and ecotoxicity. Compensation is defined as the capacity of a population to self-mitigate decreased growth, reproduction, or survival of some individuals in the population by increased growth, reproduction, or survival of the remaining individuals. The CompMech approach over the past decade has been to represent in simulation models the processes underlying daily growth, reproduction, and survival of individual fish (hence the classification of individual-based models) and then to aggregate over individuals to the population level.

The models can be used to make short-term predictions of survival, growth, habitat utilization, and consumption for critical life stages. For the longer term, the models can be used to project population abundance through time to assess the risk that abundance will fall below some threshold requiring mitigation. For stream situations, several CompMech models have been developed that couple the hydraulic simulation method of PHABSIM directly with an individual-based model of reproduction and young-of-year dynamics, thereby eliminating reliance on the habitat-based component of PHABSIM (Jager et al. 1993). The CompMech model of smallmouth bass is being used to evaluate the effects of alternative flow regimes on nest success, growth, mortality, and ultimately year class strength in a Virginia stream to identify instream flows that protect fisheries with minimum impact on hydropower production.

A model of coexisting populations of rainbow and brown trout in California is being used to evaluate alternative instream flow and temperature scenarios (Van Winkle et al. 1996). Model predictions will be compared with long-term field observations before and after experimental flow increases; numerous scientific papers are expected from this intensive study.

An individual-based model of smolt production by Chinook salmon, as part of an environmental impact statement for the Tuolumne River in California, considered the minimum stream flows necessary to ensure continuation and maintenance of the anadromous fishery (FERC 1996). That model, the Oak Ridge Chinook salmon model (ORCM), predicts annual production of salmon smolts under specified reservoir minimum releases by evaluating critical factors, including influences on upstream migration of adults, spawning and incubation of eggs, rearing of young, and predation and mortality losses during the downstream migration of smolts. Other physical habitat analyses were used to supplement the population model in evaluating benefits of alternative flow patterns. These habitat evaluations are based on data from an instream flow study; a stream temperature model was used to estimate flows needed to maintain downstream temperatures within acceptable limits for salmon.

SALMOD

The conceptual and mathematical models for the Salmonid Population Model (SALMOD) were developed for Chinook salmon in concert with a 12-year flow evaluation study in the Trinity River of California using experts on the local river system and fish species in workshop settings (Williamson et al. 1993, Bartholow et al. 1993). SALMOD was used to simulate young-of-year production, assuming that the flow schedules to be evaluated were released from Lewiston Reservoir in every year from 1976 to 1992 (regardless of observed reservoir inflow, storage, and release limitations).

The structure of SALMOD is a middle ground between a highly aggregated classical population model that tracks cohorts/size groups for a generally large area without spatial resolution, and an individual-based model that tracks individuals at a great level of detail for a generally small area. The conceptual model states that fish growth, movement, and mortality are directly related to physical hydraulic habitat and water temperature, which in turn relate to the timing and amount of regulated streamflow. Habitat capacity is characterized by the hydraulic and thermal properties of individual mesohabitats, which are the model’s spatial computational units.

Model processes include spawning (with redd superimposition), growth (including maturation), movement (freshet-induced, habitat-induced, and seasonal), and mortality (base, movement-related, and temperature-related). The model is limited to freshwater habitat for the first 9 months of life; estuarine and ocean habitats are not included. Habitat area is computed from flow/habitat area functions developed empirically. Habitat capacity for each life stage is a fixed maximum number per unit of habitat available. Thus, a maximum number of individuals for each computational unit is calculated for each time step based on streamflow and habitat type. Rearing habitat capacity is derived from empirical relations between available habitat area and number of individual fish observed.

Partly due to drought conditions, most of the flow alternatives to be evaluated did not actually occur during the flow evaluation study. When there is insufficient opportunity to directly observe and evaluate impacts of flow alternatives on fish populations,

Figure 7.39: Vegetation/ water relationship.
Soil moisture conditions often determine the plant communities in riparian areas.
Source: C. Zabawa

SALMOD can be used to simulate young-of-the-year production that may result from proposed flow schedules to be released or regulated by a control structure such as a reservoir or diversion.

Other physical habitat analyses can be used to supplement population models in evaluating benefits of alternative flow patterns. In the Trinity River Flow Study, a stream temperature model was used to estimate flows needed to maintain downstream temperatures within acceptable limits for salmon. Both the ORCM (FERC 1996) and SALMOD models concentrated on development, growth, movement, and mortality of young-of-year Chinook salmon but with different mechanistic inputs, spatial resolution, and temporal precision.

REVERSE FAST FORWARD

Review Chap. 8's information on hydroperiod/vegetation model.

Vegetation-Hydroperiod Modeling

In most cases, the dominant factor that makes the riparian zone distinct from the surrounding uplands, and the most important gradient in structuring variation within the riparian zone, is site moisture conditions, or hydroperiod (**Figure 7.39**).

Hydroperiod is defined as the depth, duration, and frequency of inundation and is a powerful determinant of what plants are likely to be found in various positions in the riparian zone. Formalizing this relation as a vegetation-hydroperiod model can provide a powerful tool for analyzing existing distributions of riparian vegetation, casting forward or backward in time to alternative distributions, and designing new distributions. The suitability of site conditions for various species of plants can be described with the same conceptual approach used to model habitat suitability for animals. The basic logic of a vegetation-hydroperiod model is straightforward. How wet a site is has a lot to do with what plants typically grow on the site. It is possible to measure how wet a site is and, more importantly, to predict how wet a site will be based on the relation of the site to a stream. From this, it is possible to estimate what vegetation is likely to occur on the site.

Components of a Vegetation-hydroperiod Model

The two basic elements of the vegetation-hydroperiod relation are the physical conditions of site moisture at various locations and the suitability of those sites for various plant species. In the simplest case of describing existing patterns, site moisture and vegetation can be directly measured at a number of locations. However, to use the vegetation-hydroperiod model to predict or design new situations, it is necessary to predict new site moisture conditions. The most useful vegetation-hydroperiod models have the following three components:

- *Characterization of the hydrology or pattern of streamflow.* This can take the form of a specific sequence of flows, a summary of how often different flows occur, such as a flow duration or flood frequency curve, or a representative flow value, such as bankfull discharge or mean annual discharge.
- *A relation between streamflow and moisture conditions at sites in the riparian zone.* This relation can be measured as the water surface elevation at a variety of discharges and summarized as a stage vs. discharge curve. It can also be calculated by a number of hydraulic models that relate water surface elevations to discharge, taking into account variables of channel geometry and roughness or resistance to flow. In some cases, differences in simple elevation above the channel bottom may serve as a reasonable approximation of differences in inundating discharge.
- *A relation between site moisture conditions and the actual or potential vegetation distribution.* This relation expresses the suitability of a site for a plant species or cover type based on the moisture conditions at the site. It can be determined by sampling the distribution of vegetation at a variety of sites with known moisture conditions and then deriving probability distributions of the likelihood of finding a plant on a site given the moisture conditions at the site. General relations are also available from the literature for many species.

The nature and complexity of these components can vary substantially and still provide a useful model. However, the components must all be expressed in consistent units and must have a domain of application that is appropriate to the questions being asked of the model (i.e., the model must be capable of changing the things that need to be changed to answer the question). In many cases, it may be possible to formulate a vegetation-hydroperiod model using representations of stream hydrology and hydraulics that have been developed for other analyses such as channel stability, fish habitat suitability, or sediment dynamics.

Identifying Non-equilibrium Conditions

In altered or degraded stream systems, current moisture conditions in the riparian zone may be dramatically unsuitable for the current, historical, or desired riparian vegetation. Several conditions can be relatively easily identified by comparing the distribution of vegetation to the distribution of vegetation suitabilities.

- The hydrology of the stream has been altered; for example, if streamflow has diminished by diversion or flood attenuation, sites in the riparian zone may be drier and no longer suitable for the historic vegetation or for current long-lived vegetation that was established under a previous hydrologic regime.

Zonation of Vegetation

There are a number of statistical procedures for estimating the frequency and magnitude of extreme events (see flood frequency analysis section of the chapter 8) and describing various aspects of hydrologic variation. Changing these flow characteristics will likely change some aspect of the distribution and abundance of organisms. Analyzing more specific biological changes generally requires defining the requirements of target species; defining requirements of their food sources, competitors, and predators; and considering how those requirements are influenced by episodic disturbance events.

- The inundating discharges of plots in the riparian zone have been altered so that streamflow no longer has the same relation to site moisture conditions; for example, levees, channel modifications, and bank treatments may have either increased or decreased the discharge required to inundate plots in the riparian zone.
- The vegetation of the riparian zone has been directly altered, for example, by clearing or planting so that the vegetation on plots no longer corresponds to the natural vegetation for which the plots are suitable.

In many degraded stream systems all of these things have happened. Understanding how the moisture conditions of plots correspond to the vegetation in the current system, as well as how they will correspond in the restored system, is an important element of formulating reasonable restoration objectives and designing a restoration plan.

Vegetation Effects of System Alterations

In a vegetation-hydroperiod model, vegetation suitability is determined by streamflow and the inundating discharges of plots in the riparian zone. The model can be used to predict effects of alteration in streamflow or the relations of streamflow to plot moisture conditions on the suitability of the riparian zone for different types of vegetation. Thus, the effects of flow alterations and changes in channel or bottomland topography proposed as part of a stream restoration plan can be examined in terms of changes in the suitability of various locations in the riparian zone for different plant species.

Flooding Tolerances of Various Plant Species

There is a large body of information on the flooding tolerances of various plant species. Summaries of this literature include Whitlow and Harris (1979) and the multivolume *Impact of Water Level Changes on Woody Riparian and Wetland Communities* (Teskey and Hinckley 1978, Walters et al. 1978, Lee and Hinckley 1982, Chapman et al. 1982). This type of information can be coupled to site moisture conditions predicted by applying discharge estimates or flood frequency analyses to the inundating discharges of sites in the riparian zone. The resulting relation can be used to describe the suitability of sites for various plant species C, e.g., relatively flood-prone sites will likely have relatively flood-tolerant plants. Inundating discharge is strongly related to relative elevation within the floodplain. Other things being equal (i.e., within a limited geographic area and with roughly equivalent hydrologic regimes), elevation relative to a representative water surface line, such as bankfull discharge or the stage at mean annual flow, can thus provide a reasonable surrogate for site moisture conditions. Locally determined vegetation suitability can then be used to determine the likely vegetation in various elevation zones.

Extreme Events and Disturbance Requirements

Temporal variability is a particularly important characteristic of many stream ecosystems. Regular seasonal differences in biological requirements are examples of temporal variability that are often incorporated into biological analyses based on habitat suitability and time series simulations. The need for episodic extreme events is easy to ignore because these are so widely perceived as destructive both of biota and of constructed river features. In reality, however, these extreme events seem to be essential to physical channel maintenance and to the long-term suitability of the riverine ecosystem for disturbance-dependent species. Cottonwood in western riparian systems is one well-understood case of a disturbance-dependent species. Cottonwood regeneration from seed is generally restricted to bare, moist sites. Creating these sites depends heavily on channel movement (meandering, narrowing, avulsion) or new flood deposits at high elevations. In some western riparian systems, channel movement and deposition tend to occur infrequently in association with floods. The same events are also responsible for destroying stands of trees. Thus maintaining good conditions for existing stands, or fixing the location of a stream's banks with structural measures, tends to reduce the regeneration potential and the long-term importance of this disturbance-dependent species in the system as a whole.